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Municipal Solid Waste: Land Disposal

Proceedings of the Fifth Annual Research Symposium

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The fifth SHWRD research symposium on land disposal and resource recovery of municipal solid waste was held at Orlando, Florida on March 26, 27, and 28, 1979. The purposes of the symposium were (1) to provide a forum for a state-of-the-art review and discussion of ongoing and recently-completed research projects dealing with the management of solid wastes; (2) to bring together people concerned with municipal solid waste management who can benefit from an exchange of ideas and information; and (3) to provide an arena for the peer review of SHWRD's overall research approach. These proceedings are a compilation of the papers presented by the symposium speakers. They are arranged in order of presentation. Volume I, land disposal, covered four primary technical areas: Pollutant Identification, Environmental Assessment, Control Technology, and Pollutant Transport. Volume II, Resource Recovery, covered five primary technical areas: Evaluation of Equipment, Unit Operations, and Processes; Economics and Impediments; Systems Analysis and Special Studies; Utilization of Recovered Materials; and Environmental Impacts of Resource Recovery.

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MUNICIPAL SOLID WASTE: LAND DISPOSAL

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at Orlando, Florida, March 26, 27, and 28, 1979
Cosponsored by the University of Central Florida
and the Solid and Hazardous Waste Research Division
U.S. Environmental Protection Agency

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FOREWORD

The Environmental Protection Agency was created because of increasing public and governmental concern about the dangers of pollution to the health and welfare of the American people. Noxious air, foul water, and spoiled land are tragic testimony to the deterioration of our natural environment. The complexity of the environment and the interplay between its components require a concentrated and integrated attack on the problem.

Research and development is the first necessary step in problem solution; it involves defining the problem, measuring its impact, and searching for solutions. The Municipal Environmental Research Laboratory develops new and improved technology and systems to prevent, treat, and manage wastewater and the solid and hazardous waste pollutant discharges from municipal and community sources; to preserve and treat public drinking water supplies; and to minimize the adverse economic, social, health and aesthetic effects of pollution. This publication is one of the products of that research -- a vital communications link between the researcher and the user community.

The Proceedings present research aimed at minimizing the impact of land disposal of solid and hazardous wastes. Solutions to specific problems are suggested.

Francis T. Mayo
Director
Municipal Environmental Research Laboratory

PREFACE

These Proceedings are intended to disseminate up-to-date information on extramural research projects dealing with the disposal of solid and hazardous wastes. These projects are funded by the Solid and Hazardous Waste Research Division (SHWRD) of the U.S. Environmental Protection Agency, Municipal Environmental Research Laboratory in Cincinnati, Ohio. Selected papers from work of other organizations were included in the symposium to identify closely-related work not included in the SHWRD program.

The papers in these Proceedings are arranged as they were presented at the symposium. There were two concurrent programs, one primarily for land disposal and the other primarily for resource recovery. Each program had multiple sessions. There was an introductory session. Each of the five sessions includes papers dealing with major areas of interest for those involved in solid and hazardous waste management and research. Each program, land disposal and resource recovery, is printed in a separate volume.

The papers are printed here basically as received from the symposium authors. They do not necessarily reflect the policies and opinions of the U.S. Environmental Protection Agency or the University of Central Florida. Hopefully, these Proceedings will prove useful and beneficial to the scientific and engineering community as a current reference on land disposal and resource recovery.

ABSTRACT

The fifth SHWRD research symposium on land disposal and resource recovery of municipal solid waste was held at Orlando, Florida on March 26, 27, and 28, 1979. The purposes of the symposium were (1) to provide a forum for a state-of-the-art review and discussion of ongoing and recently completed research projects dealing with the management of solid wastes; (2) to bring together prople concerned with municipal solid waste management who can benefit from an exchange of ideas and information; and (3) to provide an arena for the peer review of SHWRD's overall research approach. These proceedings are a compilation of the papers presented by the symposium speakers. They are arranged in order of presentation. Volume I, Land Disposal, covered four primary technical areas: Pollutant Identification; Environmental Assessment; Control Technology; and Pollutant Transport. Volume II, Resource Recovery, covered five primary technical areas: Evaluation of Equipment, Unit Operations, and Processes; Economics and Impediments; Systems Analysis and Special Studies; Utilization of Recovered Materials; and Environmental Impacts of Resource Recovery.

This report was submitted in fulfillment of Research Grant No. 806198 by the University of Central Florida under the sponsorship of the U.S. Environmental Protection Agency. This report covers the period from October 1978 to July 1979, and work was completed as of July 1979.

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PROGRAMS IN LAND DISPOSAL AND RESOURCE RECOVERY AND CONSERVATION

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ABSTRACT

This paper describes EPA's programs in land disposal and resource recovery and conservation. The programs are authorized by the Resource Conservation and Recovery Act (RCRA) of 1976, which also authorizes the hazardous waste programs carried out by the Agency. RCRA is implemented by EPA's Office of Solid Waste. The land disposal and resource recovery and conservation programs include:

1. development of regulations and guidelines,
2. technology demonstrations and evaluations,
3. studies and information development, and
4. technical and financial assistance.

The basic purpose and objectives of these activities are summarized and expected accomplishments and milestones for the next year are presented.

INTRODUCTION

Programs of the U.S. Environmental Protection Agency, Office of Solid Waste (OSW) are carried out under the authority of the Resource Conservation and Recovery Act of 1976 (RCRA). The basic objectives of this Act are to promote the application of improved solid waste management in order to protect public health and the environment and to conserve valuable material and energy resources. Based on the requirements and priorities in the Act, OSW has developed programs in these areas:

1. hazardous waste management,
2. land disposal, and
3. resource recovery and conservation.

Since the subjects of this Symposium are land disposal and resource recovery, this paper will focus only on those activities. First, specific projects on land disposal and resource recovery will be discussed, followed by a discussion of the technical and financial assistance available to help build State and local government programs.

LAND DISPOSAL

Land disposal activities can be grouped into two categories:

1. development of regulations and guidelines, and
2. technology demonstrations and technical information.

Regulations and Guidelines

The keystone to the disposal provisions of RCRA are the Criteria for Classification of Solid Waste Disposal Facilities (the Disposal Criteria) called for by Section 4004 of the Act. The Disposal Criteria will establish national standards for solid waste disposal by distinguishing facilities that pose no reasonable probability of adverse effects on health or the environment. Facilities not complying with these criteria are termed "open dumps" and are prohibited by the Act. The Disposal Criteria were published in proposed form in the Federal Register on February 6, 1978. Final promulgation is expected in mid-1979.

Section 4005 of RCRA requires EPA to publish an inventory of all disposal facilities in the United States that are open dumps, i.e. facilities that do not comply with the Disposal Criteria. Any facility found to be an open dump must be closed or upgraded according to a State-established compliance schedule. Financial assistance will be provided to State solid waste programs to evaluate facilities for purposes of the open dump inventory. A site classification manual is being developed which will provide a practical interpretation of the Disposal Criteria and will be useful to the States in evaluating facilities for purposes of the open dump inventory. Also another manual is being developed describing procedures and techniques for closing or upgrading open dumps.

RCRA calls for publication of the open-dump inventory one year after final promulgation of the Disposal Criteria. However, due to the large number of facilities to be evaluated and the complexity of such evaluations, the inventory will be published in annual installments. An arrangement has been made with the Bureau of Census to handle the data processing and actual inventory publication.

For the most part the Disposal Criteria do not specify particular technologies or operational approaches but instead establish performance levels as necessary to protect public health and the environment. This was done to allow different technologies and operations to be used as the local situation dictated and to provide opportunity for new and innovative approaches. However, RCRA does call upon EPA to provide guidance on technologies.

Section 1008 directs the Agency to develop and publish suggested guidelines that provide a technical and economic description of various solid waste management practices (including operating practices). One such guideline describing the practice of landfilling is under development at this time. This guideline will discuss design and operation of a landfill in order to protect public health and the environment and will recommend practices for leachate control, gas migration control and groundwater monitoring. A working draft of the Landfill Guideline was circulated last year and a proposed version should be published in the Federal Register for public comment by March of 1979.

Other guidelines to be developed in the future will cover landfarming of solid waste and surface impoundment disposal. In addition, preparatory work is underway to develop guidelines on the management of two special waste categories: coal-fired electric utility wastes and mining wastes. In cooperation with the Office of Research and Development programs are being initiated to evaluate facilities that dispose of fly ash, bottom ash, and scrubber sludge from coal-fired utilities and wastes from metals and phosphate mining and beneficiation processes and uranium mining. Thirty to forty facilities will be evaluated over a period of two years in order to assess practices for managing such wastes.

Finally a guideline is being developed for the disposal and utilization of wastewater treatment plant sludge under the combined authorities of RCRA and Section 405 of the Clean Water Act. This guideline will discuss practices for landfilling sewage sludge, use of sludge as a soil conditioner on agricultural lands, thermal processing of sludge for energy recovery, and sludge composting or heat drying to produce a bulk or bagged product. These guidelines will be proposed in mid-1979 with final promulgation expected in 1980.

Section 405 of the Clean Water Act requires compliance with such guidelines by operators of publicly owned wastewater treatment works and it is expected that these guidelines will be administered through the National Pollution Discharge Elimination System permit process.

Technology Demonstrations and Technical Information

The Office of Solid Waste has several projects underway to demonstrate and evaluate improved techniques for disposing of solid waste on land.

In Tullytown, Pennsylvania, leachate from an asphalt-lined landfill is collected and treated by a number of physical and biological methods. The leachate treatment plant has been operating for several years and a number of EPA reports have described its operation. The final project report was received in early 1979 and should be publicly available shortly. The treated leachate is discharged to a nearby river in conformance with standards established by the State.

In 1979, full-scale operation of another leachate treatment facility in Enfield, Connecticut is expected to begin. This facility, based upon research conducted by the EPA, Office of Research and Development and the University of Illinois. uses an anaerobic filter to treat leachate. One of the features of this system is the low volume of sludge produced. The final report on this project is expected in mid-1980.

A full-scale lined landfill demonstration is underway in Lycoming County, Pennsylvania. After several years in development, a landfill employing a PVC liner to collect leachate began to receive wastes in early 1978. A report will be available in 1979 covering among other things, procedures for installing a liner sheet covering a large area. Two other lined landfill demonstrations in Northern Tier, Pennsylvania are expected to start in 1979.

In Bangor, Maine the Office of Solid Waste is participating with the City in demonstrating a process for composting sewage sludge. This project is based upon the aerated-pile process for composting sludge developed by the U.S. Department of Agriculture in Beltsville, Maryland. This demonstration represents the first on-line application of this process in a small community. It has proven very successful and composted sewage sludge has been sold and used as a soil conditioner for over a year. An interim report was published in 1977 and a final report is due in 1979.

In Mountain View, California a very innovative project involving the recovery of methane from a landfill is underway. In a joint effort involving the City, the Pacific Gas and Electric Company and EPA, methane gas is pumped from a landfill, cleaned up, and injected into a nearby pipeline supplying gas to residential users. This project not only prevents the safety problems that could result from the migration of gas from the landfill, but also recovers and utilizes a valuable resource. The facility has been operational for nearly a year, interim reports are available, and a final report is expected in mid-1979.

The Office of Solid Waste also prepares a number of technical reports on various subjects pertaining to land disposal. One such report that was very well received was a procedures manual for monitoring leachate and groundwater around disposal facilities. We also have an ongoing effort to monitor groundwater at a number of disposal sites both to improve monitoring techniques and gain a better understanding of leachate and groundwater interaction. Other technical reports under development include an assessment of methods to prevent and control migration of toxic or explosive gases at disposal sites and an evaluation of procedures to prevent hazards to aircraft from birds attracted to disposal facilities.

The Office of Solid Waste also participates in the review and evaluation of the wide range of research and development projects in the land disposal area conducted by the Office of Research and Development and described in other papers at this Symposium.

RESOURCE RECOVERY AND CONSERVATION

Resource recovery and conservation activities will be grouped into three categories for discussion:

1. activities of the Resource Conservation Committee
2. guidelines for Federal agencies, and
3. demonstrations, evaluations and studies.

Resource Conservation Committee

The Resource Conservation Committee was established by Section 8002(j) of the Act. It is a Cabinet-level committee chaired by the Deputy Administrator of EPA and supported by OSW staff. The Committee is required to conduct "a full and complete investigation and study of all aspects of the economic, social, and environmental consequences of resource conservation." To obtain public input for these studies, the Committee held several hearings on beverage container deposits, solid waste disposal charges, the effect of existing Federal tax and transportation policies on the use of virgin and secondary materials, recycling and resource recovery subsidies, deposits or bounties on durable goods, local user fees, litter taxes, severance taxes, and product regulation.

The first resource conservation issue taken up by the Committee was beverage container deposits. In its January 1978 report to the President and Congress, the Committee presented a recommended design for beverage container deposit legislation. The Committee did not recommend that such legislation be passed, however; a decision on this issue was postponed until further analyses were completed.

In its July 1978 report to the President and Congress, the Committee submitted preliminary analyses of solid waste disposal charges. Work is continuing on beverage container deposits and solid waste disposal charges and the results of these and other analyses will be presented in the Committee's final report along with recommendations. This report is scheduled for completion in March 1979.

Guidelines for Federal Agencies

On April 23, 1976, EPA issued Guidelines for Source Separation for Materials Recovery. These guidelines require the recovery of highgrade paper, newsprint, and corrugated boxes from Federal facilities. As a result, 175,000 Federal employees in 135 Federal facilities are participating in the paper recycling programs. It is expected that another 100,000 employees will be in the program by October 1979. OSW has worked with the General Services Administration in establishing contract procedures for the sale of the paper and in developing an educational program for training Federal employees in procedures for recycling office paper. To date, over 2,000 tons of paper have been recycled under the guidelines with a return to the Federal Treasury of over $125,000. In addition, at least 15 States and hundreds of local governments have adopted the guidelines.

On September 21, 1976, EPA issued Guidelines for Beverage Containers. These guidelines require that a refundable 5-cent deposit be placed on all containers for beer and soft drinks sold at Federal facilities. The deposit is intended to encourage the return of containers for either refilling or recycling. The Department of Defense concluded a test of the guidelines at 10 military bases in June 1978 and will submit its plans based on this test in early 1979. The General Services Administration also conducted a test of the guidelines in 8 of its 10 regions. To date, 13 Federal agencies are implementing the guidelines. The OSW monitors agency compliance with the guidelines and provides assistance upon request.

On September 21, 1978, EPA issued Resource Recovery Facilities Guidelines. These guidelines contain requirements and recommended procedures for Federal agencies regarding establishment and use of resource recovery facilities. To date, some degree of implementation has been reported by Federal agencies in nine metropolitan areas.

Procurement Guidelines are currently being developed under Section 6002 of RCRA. All agencies procuring with Federal funds (including State and local governments, grantees, and contractors as well as Federal agencies) must "procure items composed of the highest percentage of recovered materials practicable." The guidelines for recommended procurement practices will contain information regarding suppliers, demand, price, delivery time, performance, and certification techniques.

OSW is now gathering data on products purchased by the government that may have high potential in the use of waste materials. Studies include:

1) The use of fly ash and blast furnace slag in cement and concrete. 2) The use of wastepaper in insulation and board; waste rubber in asphalt pavements; and waste glass in bricks, asphalt, and concrete. 3) The use of wastepaper and other secondary fibrous materials in paper products and 4) The use of composted sewage sludge used as a soil conditioner and low-grade fertilizer.

The first guideline, scheduled to be proposed in April 1979, will be on the use of fly ash and blast furnace slag in cement and concrete.

Demonstrations, Evaluations and Studies

The OSW is involved in the demonstration and evaluation of the technical, economic and environmental performance of a number of resource recovery systems.

Two of the early demonstration projects developed technologies that are being replicated in a number of communities. The first, in Franklin, Ohio, determined the feasibility of using the wet-pulping method of separating mixed municipal solid waste into organic and inorganic fractions. Follow-on applications of this technology will use the organic fraction of the solid waste as a fuel.

The second demonstration, in St. Louis, Missouri, developed the technology of recovering a portion of the organic fraction of the mixed municipal waste stream for use as a supplement to coal in large boilers. This technology is being replicated in 10 locations. An extension of this concept that includes utilization of sewage sludge is being demonstrated under a grant to the State of Delaware.

Two systems for recovering energy from mixed municipal solid waste through pyrolysis were demonstrated in Baltimore, Maryland, and San Diego County, California. The Baltimore plant, which produces steam by combusting the pyrolysis gases, has had numerous mechanical and air pollution problems and is presently undergoing an extensive modification program. After the modifications are completed, it is anticipated that the process will run in an economically and environmentally sound manner. The San Diego project also experienced difficulties and is currently not in operation. Although it is doubtful that either of these systems will be replicated in toto, much was learned concerning the pyrolysis of solid waste, and this information should prove useful to others considering this process.

Six projects to develop source separation systems are now nearing completion. Projects in Marblehead and Somerville, Massachusetts, were started three years ago to determine the feasibility of separate collection techniques to recover several materials from municipal waste streams. These two programs have shown that it is possible to recover between 25 and 30 percent of the residential waste stream in suburban communities. Approximately 15 to 20 other multimaterial collection programs have begun in the New England area as a result of the increased municipal and industry interest created by the Marblehead/Somerville projects. A final report presenting the results of this project will be issued in 1979.

Four projects were started two years ago on: multimaterial collection through private contract; multimaterial collection from apartment buildings; source separation and materials marketing for low-density rural areas; and the use of handicapped laborers for processing materials. Final reports will be issued in 1979.

In addition to EPA-supported demonstrations of resource recovery systems, OSW is also conducting evaluations of other commercial-scale resource recovery facilities: refuse-derived fuel plants in Lane County, Oregon, and Chicago, Illinois; small modular incinerators with heat recovery; waterwall combustion units in Europe; a plant for codisposal of sewage sludge and municipal solid waste in Duluth; and a pyrolysis system in Frankfort, Germany. Evaluations of system components such as air classifiers and shredders are also underway.

In cooperation with the Office of Research and Development, OSW is conducting studies on resource recovery as required by section 8002 of the Act on the subjects of: 1) compatibility of source separation and mixed waste recovery, 2) small-scale technology, 3) research priorities for resource recovery and 4) tire, glass and plastics recovery.

FINANCIAL AND TECHNICAL ASSISTANCE TO STATE AND LOCAL GOVERNMENTS

The Resource Conservation and Recovery Act places the primary responsibility of improving solid waste management on State and local governments. The Act provides for a number of financial and technical assistance programs for assisting in that effort.

Financial assistance is available to develop and implement comprehensive State solid waste management plans under Subtitle D of the Act. Guidelines to assist in the development and implementation of State solid waste management plans were published in proposed form in August 1978 and will be finalized in June 1979. These plans should provide for (1) regulatory programs for environmentally sound disposal of all other solid wastes, (2) State programs to encourage resource recovery and conservation, and

(3) planning for necessary disposal and recovery facilities. In fiscal year 1979, $11.2 million was available for these purposes.

A new financial assistance program resulting from the President's Urban Policy Message will help communities to adequately assess the feasibility of resource recovery projects and obtain sufficient consultation and staff for the preparatory steps in implementation. This program is scheduled to begin in fiscal year 1979 and $15 million has been appropriated for the first year. Urban areas will be eligible for assistance for "front-end" steps preceding actual design and construction of resource recovery projects. This includes not only mixed waste resource recovery processing plants but also projects involving source separation (i.e., separation at the place of generation) of waste materials for recycling. Eligibility for funding is not limited to large cities, but a major portion of the funds will probably go to jurisdictions of at least 50,000 population. The aid will go primarily to agencies with clear responsibility for implementation as designated in the State planning process under Subtitle D. The first set of applications were received in December 1978 and awards are expected to be made in March or April 1979.

The Act also provides for technical assistance through the Technical Assistance Panels Program. Each EPA regional office has available a panel of expert consultants capable of assisting in all areas of solid waste management. Grants have been provided to a number of public interest groups to make State and local government officials available to provide assistance as well. These panels are available to provide technical assistance free of charge to State and local governments and Federal agencies upon request. Twenty percent of the funds appropriated under the general authorization of the Act are used for this purpose.

The OSW has developed a seminar series directed mainly at municipal officials. A first set of seminars presented alternatives and major issues in the implementation of resource recovery. In 1978 the seminar was conducted in six cities; approximately 1,000 people attended. The seminar program will continue into 1979 and will be expanded to cover other solid waste management issues as well.

The Resource Conservation and Recovery Act provides an opportunity for significant advancements in solid waste management nationwide. The regulations and guidelines provide momentum and impetus for improved practices. Technology demonstrations, evaluations and studies will help provide necessary technical, economic and environmental performance information so that better decisions can be made. Financial and technical assistance programs will provide additional resources to State and local governments responsible for implementation. The Office of Solid Waste programs in these three areas are an attempt to provide a balanced approach to this nationwide effort.

USEPA REGION 4 SOLID WASTE MANAGEMENT ACTIVITIES

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In the beginning -- in this instance the late 1960s -- solid waste management in Region 4 was at best completely inadequate even by standards prevailing at that time. Before getting into an update on regional solid waste management activities we must look at our beginning.

Prior to the passage of the Solid Waste Disposal Act of 1965, there did not exist an identifiable state solid waste program in the region. A survey of land disposal sites in the late 1960s indicated over 2500 dumps and fewer than 30 sanitary landfills which could meet the very loose requirements that prevailed at that time. By 1971, total state solid waste program positions were 42.

State solid waste laws were enacted, viable programs established, regulations promulgated, and state plans developed. Inspection and technical assistance programs were put into action.

By 1977, over 1000 land disposals met or exceeded the minimum requirements to be termed a sanitary landfill and permitted or approved by the state programs. State program personnel now exceeds 225 with increases scheduled to take on the additional duties associated with hazardous waste and resource recovery.

While maintaining the successful level of solid waste management attained, a combined federal, state and local governmental effort is underway to correct and eliminate existing and potential environmental hazards from old disposal practices. New and environmentally sound hazardous waste processing and disposal facilities are being designed and constructed. Some are now in operation.

Resource recovery facilities are in operation, under construction, and being planned. Under the President's Urban Policy, which provides financial assistance for Resource Recovery Project Development, more than thirty applications requesting over $3.5 million in Federal assistance were received from this region. The Department of Energy has solid waste demonstration projects in Alabama, Florida and Georgia.

EPA funding of state solid waste programs in Fiscal Year 1979 will exceed four million dollars. Technical assistance through Section 2003 of the Resource Conservation and Recovery Act has been delivered to more than 30 local governments at an approximate cost of $200,000. This was provided at no cost to the clients, i.e. local governments.

In the 1980s, we expect to achieve a level of solid waste management, hazardous waste control and resource recovery that should meet our immediate needs and set examples for the rest of the Country.

CURRENT RESEARCH ON LAND DISPOSAL OF MUNICIPAL SOLID WASTES

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ABSTRACT

The Solid and Hazardous Waste Research Division (SHWRD), Municipal Environmental Research Laboratory, U.S. Environmental Protection Agency, in Cincinnati, Ohio, has responsibility for research in the areas of municipal solid and hazardous waste management, including both disposal and processing. This research is being directed towards new and improved systems of municipal solid and hazardous waste management, development of technology, determination of environmental effects, and collection of data necessary for the establishment of disposal and processing guidelines.

Division activities in the area of municipal solid waste research have related to storage, collection, transport, processing, resource recovery, and disposal. Recent emphasis on energy has resulted in an expansion of the waste-as-fuels program.

The current municipal solid waste diposal research program has been divided into three general areas: (1) Pollutant Predictions for Current Landfill Techniques, (2) Alternatives to Current Landfill Disposal Techniques, and (3) Remedial Action for Minimizing Pollutants from Unacceptable Sites.

The research activities currently funded under these three general areas have been classified into seven categories shown below:

1. Waste Characterization/Decomposition
2. Pollutant Transport
3. Pollutant Control/Treatment
4. Co-disposal
5. Environmental Assessment
6. Remedial Action
7. Landfill Alternatives

INTRODUCTION

Increasing amounts of waste residuals are being directed to the land for disposal in landfills. At the same time, there is increasing evidence of environmental damage resulting from improper operation. The burden of operating landfills and coping with any resulting damages falls most heavily on municipalities and other local government agencies. Their problems are complex, involving legislation, economics, and public attitudes as well as technology; additionally, comprehensive information on landfilling techniques and protection of the local environment is not readily available.

Current estimates indicate that 144 million tons (as generated with moisture) of municipal wastes and 260 million tons (dry) of industrial wastes are disposed of to the land. It is projected that approximately 197 million tons of municipal solid wastes will be generated in the United States in 1990. A survey of national solid waste management practices conducted in 1968 by the Federal government through cognizent state agencies indicated less than 6 percent of 6000 land disposal sites surveyed could be considered sanitary landfills, based on very modest criteria not including a re-evaluation of groundwater pollution. More recent surveys (in 1975 and 1977) indicated a 25 percent increase over a 2-year period in the number of disposal sites considered to be sanitary landfills (5740 of 15,821 sites in 1976). The total number of municipal solid waste disposal sites (18,500 sites in 1977) appears to have increased greatly since 1968, but this can be explained, in part of better record keeping by state agencies and/or that environmental enactments have both severely restricted refuse disposal by burning or dumping at sea and greatly increased the amount of solid waste generated as residues from air and water cleaning operations. The most pernicious effect of unsound disposal is the contamination of groundwater by leachate; about half of the United States domestic water supply is from groundwater. Groundwater contamination is usually discovered long after the damage is done and too late for corrective measures.

The municipal waste disposal program initiated by SHWRD was designed to document and evaluate the potential adverse environmental and public health effects that could result from application of waste disposal methods without proper precautions for leachate and gas management. The information thus obtained will provide the necessary data for the establishment of guidelines for communities to develop economical and environmentally safe municipal waste disposal management systems.

Specifically, in the area of landfill disposal techniques, a comprehensive data base on the characteristics of municipal and hazardous wastes will be developed to assess pollutants within a waste residual, pollutant release from waste residuals in the form of leachates and gas, decomposition rates under varying moisture regimes, the migration and attenuation of pollutants, and the resultant environmental damages. In addition, liner materials, both natural and synthetic, and chemical stabilization techniques for controlling leachate movement will be investigated along with treatment of the collected leachates. Research efforts will be conducted primarily through laboratory and pilot studies with some field testing of laboratory-based results.

In the area of alternative land disposal methods, technical and environmental data will be obtained to provide a basis for logical engineering decisions on viable environmentally sound methods other than landfill trench method.

In the area of remedial action for preventing pollutant generation from unacceptable landfill sites, engineering feasibility and design plans will be developed and tested in field verification studies.

This research strategy, encompassing state-of-the-art documents, laboratory analysis, bench and pilot studies, and full-scale field verification studies, is at various stages of implementation, but over the next 5 years the research reports developed will be compiled as criteria and guidance documents for user communities. Also, the waste disposal research program will develop and compile a research criteria data base for use in the development of guidelines and standards for waste residual disposal to the land as mandated by the recently enacted legislation entitled "Resource Conservation and Recovery Act of 1976" (RCRA).

WASTE CHARACTERIZATION/DECOMPOSITION

Studies in this area involve collecting composition data on municipal and hazardous wastes from individual waste residuals and landfill disposal sites. Sampling techniques, analytical methods/procedures, and waste compatibility and waste decomposition information will be developed for implementing better disposal practices and waste management.

The objectives of this research activity are to (1) quantify the gas and leachate production from current best-practice, sanitary landfilling and (2) modify the landfill method to reduce the environmental impact of gas and leachate production in a positive and predictable manner. These objectives are to be achieved by construction and long-term monitoring of typical and simulated landfill cells and investigation, development, and optimization of those factors that control gas and leachate production. Results are expected only after long-term monitoring, due to the extremely slow reaction rates.

Several previous reviews of these efforts have been presented. See Schomaker, N.B. and Roulier, M.H., Current EPA Research Activities in Solid Waste Management: Management of Gas and Leachate in Landfills: Proceedings of the Third Annual Municipal Solid Waste Research Symposium, University of Missouri, EPA-600/9-77-026; and Schomaker, N.B., Current Research on Land Disposal of Hazardous Wastes: Land Disposal of Hazardous Wastes - Proceedings of the Fourth Annual Research Symposium, EPA-600/9-78-016.

Standard Analytical Techniques

Analysis of the contaminants within a waste leachate sample is difficult due to interfering agents. Existing instrumentation functions well in the analysis of simple mixtures at low concentrations but interference problems can be encountered for complex mixtures at high concentrations (1 percent by weight and greater). In this range the sample cannot always be analyzed directly and commonly must be diluted and/or analyzed by the method of standard additions. Options are the development of standard procedures for diluting and accounting for errors introduced thereby or the development of instrumentation capable of accurate, direct measurements at high concentrations in the presence of potential multiple intereferences. Existing USEPA procedures for water and wastewaters are often not applicable. Analytical procedures are being developed on an as-needed basis as part of the SHWRD projects. However, most of this work is specific to the wastes being studied and separate efforts were required to insure that more general procedures/equipment would be developed. A compilation of analytical techniques[1]* used for contaminant analysis has been published in a report entitled Compilation of Methodology Used for Measuring Pollution Parameters of Sanitary Landfill Leachates, EPA-600/3-75-011, October 1975, and SHWRD is currently conducting a collaborative testing study[2] on leachate analyses. In this study, leachate samples were sent to 51 laboratories for analysis of specific parameters. The results obtained have provided information on detection limits and precision for contaminants in leachate, using currently accepted methods developed for water and wastewater.

A second effort[2] that has been recently initiated will determine which sampling and preservation techniques should be accepted as standards for groundwater sampling. Other objectives of the study are to determine if current sampling methods produce samples representative of water contained in the aquifer being monitored and if groundwater samples collected in the field can be treated on location or if laboratory treatment is required. Six landfill monitoring wells will be studied using four different pumping techniques and thirteen different sample preservation procedures.

A third effort[2], recently initiated, is an in-house activity to determine the capability of existing analytical procedures to quantitate priority pollutants in various waste landfill leachates. Preliminary efforts will obtain leachates from four landfill sites, two municipal solid wastes and two hazardous waste sites, to determine specific priority pollutants. Data will be analyzed to resolve analytical methodology applicable to leachate samples.

Standard Leaching Tests

In studying the potential environmental impact of contaminants, a standard leaching test is needed to assess contaminant release from a waste. Such a test must provide information on the initial release of contaminants from a waste contacted not only by water but also by other solvents that could be introduced in disposal. Additionally, such a test must provide some estimate of the behavior of the waste under extended leaching. Experience from ongoing SHWRD projects indicates that some wastes may initially release only small amounts of contaminants, but, under extended leaching, will release much higher concentrations. Such leaching behavior has an impact on disposal regulation and on management of a disposal site, and information on this behavior must be obtained and used in classifying a waste. The Office of Solid Waste (OSW) has funded the Industrial Environmental

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*Superscript numbers refer to the project officers, listed immediately following this paper, who can be contacted for additional information.

Research Laboratory (IERL), USEPA project[3] to examine this background area and develop procedures for determining whether a waste contains a significant level of toxic contaminants and whether a waste will release such contaminants under a variety of leaching conditions.

Validation of a Standard Leaching Test (SLT) is an effort currently underway, funded in part by SHWRD.[2] The passage of the RCRA on October 21, 1976, imposing time restraints, necessitated developing an Interim Standard Leaching Test (ISLT). Existing leaching tests were evaluated for those elements that were of special benefit to the development of an SLT. This information has been published in a report entitled Compilation and Evaluation of Leaching Test Methods, EPA-600/2-78-095, May 1978. Three candidate SLT's were chosen for further testing. This additional effort is being published in two reports entitled Comparison of Three Wastes Leaching Tests, Executive Summary EPA-600/8-79-001 and the Background Data Report. Also, an Environmental Research Brief[2] dedicated to the rapid dissemination of information relative to the solid waste "Extraction Procedure" (EP) portion of Section 3001 of RCRA is being published.

A second effort[2] identified as the "Extraction Procedure" (EP) test will establish a data base on toxicity of leachates from solid wastes and establish recommended toxicity test protocols in order to establish criteria defining acute and/or chronic hazardous wastes. Work has been initiated on four waste categories: municipal refuse, Chicago sewage sludge, arsenic sludge and power industry residuals.

Waste Leachability

The characteristics of leachates from municipal refuse and mixtures of municipal refuse and selected industrial and municipal sludges are being studied in several different projects. Leaching study results[3] for 437 tons of municipal refuse are being compared to results of leaching from 117-ton and 3-ton experimental landfills located at the Boone County Field Site (BCFS) of USEPA. This effort is being published in an interim report entitled, Boone County Field Site Report: Test Cells 2A, 2B, 2C, 2D. The viral and bacterial efforts at BCFS have been reported in the News of Environmental Research in Cincinnati, "Survival of Fecal Coliforms and Fecal Streptococci in a Sanitary Landfill" (April 12, 1974) and "Poliovirus and Bacterial Indicators of Fecal Pollution in Landfill Leachates" (January 31, 1975). More information can be obtained from the Boone County Field Site Interim Report.

A strong effort[3] performed under contract at the Center Hill Facility of USEPA, involves comparing the characteristics of leachates obtained from 3-ton experimental landfills containing municipal refuse and selected sludges of municipal and industiral origin. The effort is being published in a report entitled Pilot Scale Evaluation of Sanitary Landfilled Municipal and Industrial Wastes.

A third effort[3] currently going on will determine the health and environmental significances of the persistence of fecal streptococci found in landfilled municipal refuse. One phase of the study will verify microbial analytical methods and determine the presence of study organisms in a variety of leachates. The second phase will study the relationship between the extent of waste decomposition and the microbial population dynamics.

Waste Decomposition

Waste decomposition data are being obtained from several ongoing efforts. One study[4] is a modification of the landfill method to accelerate waste decomposition in a predictable manner. This study was successfully performed on a laboratory scale and a report entitled Sanitary Landfill Stabilization with Leachate Recycle and Residual Treatment, EPA-600/2-75-043, October 1973, has been published. Leachate recirculation with pH control results in waste decomposition in a time period as short as 6 months. Field application will probably yield a 1- or 2-year decomposition period due to anticipated leachate distribution problems.

A second effort[4] involves the elucidation of the role of moisture regime (different net infiltration conditions). This study is being performed on a laboratory scale. It will yield valuable information with respect to the kinetics of waste decomposition, including CH4 volumes and production rates. Temperature, density and moisture all appear to affect gas volumes produced.

A third effort[4] involves the effect of different waste processing techniques on gas and leachate production and duration during waste decomposition. Raw refuse, shredded refuse, and baled refuse are being investigated in a simulated landfill environment. Interim results of this effort have been previously discussed at the Engineering Foundation and ASCE Conference: Land Application of Residual Materials, held in Easton, Maryland, September 26 to October 1, 1976, as "A Study of Gas and Leachate Production from Baled and Shredded Municipal Solid Wastes." This effort is being published in an EPA report entitled Effect of Processing on MSW Decomposition.

POLLUTANT TRANSPORT

Pollutant transport studies consider the release of pollutants in liquid and gaseous forms from various municipal and hazardous wastes and the subsequent movement and fate of these pollutants in soils adjacent to disposal sites. Although the potential for damage in general can be demonstrated, migration patterns of contaminants and consequent damages that would result from unrestricted landfilling at specific sites cannot be accurately predicted. The ability to predict must be developed in order to justify the requirement for changes in the design and operation of disposal sites, particularly for any restriction of co-disposal of municipal and industrial waste. Both laboratory and field verification studies at selected sites are being performed to assess the potential for groundwater contamination. The studies will provide the information required to (1) select land disposal sites that will naturally limit release of pollutants to the air and water and (2) make rational assessments of the need for and cost-benefit aspects of leachate and gas control technology.

The overall objective of this research activity is to develop procedures for using soil as a predictable attenuation medium for pollutants. Not all pollutants are attenuated by soil, and, in some cases, the process is one of delay so that the pollutant is diluted in other parts of the environment. Consequently, a significant number of the research projects funded by SHWRD are focused on understanding the process and predicting the extent of migration of contaminants (chiefly heavy metals) from land disposal sites.

These pollutant migration studies are being performed simultaneously in the areas of hazardous wastes and municipal refuse. Several previous discussions of these efforts have been presented. See Roulier, M.H., Attenuation of Leachate Pollutants by Soils, presented at the Management of Gas and Leachate in Landfills: Proceedings of the Third Annual Municipal Solid Waste Research Symposium, March 14-16, 1977, University of Missouri.

Bibliography and State of the Art

A bibliography[5] is being published for distribution by NTIS only. This document entitled Transport and Transformation of Hazardous Substances in Soil: A Bibliography is the result of a search of literature for 1970 through 1974 and includes information on the transport, transformation, and soil retention of arsenic, asbestos, beryllium, cadmium, chromium, copper, cyanide, lead, mercury, selenium, zinc, halogenated hydrocarbons, pesticides, and other hazardous substances. The bibliography has been divided into two volumes to facilitate its use; the pesticides citations have been placed in a separate volume. The bibliography is being published through NTIS only because most of the information has been included in the report Movement of Selected Metals, Asbestos, and Cyanide in Soils: Applications to Waste Disposal Problems described below.

A state-of-the-art document[5] on migration through soil of potentially hazardous pollutants contained in leachates from waste materials has been published, Movement of Selected Metals, Asbestos, and Cyanide in Soils: Applications to Waste Disposal Problems, EPA-600/2-77-022, April 1977. The document presents a critical review of the literature pertinent to biological, chemical, and physical reactions, and mechanisms of attenuation (decrease in the maximum concentration for some fixed time as distance traveled) of the selected elements arsenic, beryllium, cadmium, chromium, copper, iron, mercury, lead, selenium, and zinc, together with asbestos and cyanide, in soil systems.

Controlled Lab Studies

The initial effort[5] examined the factors that attenuate contaminants (limit contaminant transport) in leachate from municipal solid waste landfills. Although work was strongly oriented towards problems of disposal of strictly municipal wastes, the impact of co-disposal of municipal and hazardous wastes were also considered. The project studied contaminants normally present in leachates from municipal solid waste landfills and with contaminants that are introduced or increased in concentration by co-disposal of hazardous wastes. These contaminants are: arsenic, beryllium, cadmium, chromium, copper, cyanide, iron, mercury, lead, nickel, selenium, vanadium, and zinc. The general approach was to pass municipal leachate through columns of well characterized whole soils maintained in a saturated anaerobic state. The typical municipal refuse leachate was spiked with high concentrations of metal salts to achieve a nominal concentration of 100 mg/l. The most significant factors in contaminant removal were then inferred from correlation of observed migration rates and known soil and contaminant characteristics. This effort contributed to the development of a computer simulation model for predicting trace element attenuation in soils. Modeling efforts to date have been hindered by the complexity of soil-leachate chemistry. The results of this effort have been released in a final report entitled, Investigation of Landfill Leachate Pollutant Attenuation by Soils, EPA-600/2-78-158, August 1978.

The second effort[5] in this area studied the removal of contaminants from landfill leachates by soil clay minerals. Columns were packed with mixtures of quartz sand and nearly pure clay minerals. The leaching fluid consisted of typical municipal refuse leachate without metal salt additives. The general approach to this effort was similar to that described in the preceding effort except that (1) both sterilized and unsterilized leachates were utilized to examine the effect of microbial activity on hydraulic conductivity and (2) extensive batch studies of the sorption of metals from leachate by clay minerals were conducted. Final results of this effort have been reported in a publication entitled, Attenuation of Pollutants in Municipal Landfill Leachate by Clay Minerals, EPA-600/2-78-157, August 1978.

The third effort[4] relates to modeling movement in soil of the landfill gases, carbon dioxide and methane. The modeling movement has been verified under laboratory conditions. This effort has not focused on the impact of gases on groundwater, but considers groundwater as a sink for carbon dioxide. Results evaluate the efficiency of vertical chimney vents, barriers, and forced convection systems.

A fourth effort[5] relates to the use of large-scale, hydrologic simulation modeling as one method of predicting contaminant movement at disposal sites. The two-dimensional model that was used successfully to study a chromium contamination problem is being developed into a three-dimensional model and will be tested on a well-monitored landfill where contaminant movement has already taken place. Although this type of model presently needs a substantial amount of input data, it appears promising for determining contaminant transport properties of field soils and, eventually, predicting contaminant movement using a limited amount of data.

A fifth effort[5] is using an existing mathematical model for solute movement in soil to develop a simplified empirical method for predicting metal movement in soil. It is intended that this method be oriented toward a larger number of users than the large-scale model described above. The approach is to measure metal movement rates in a number of well-characterized soils from throughout the U.S. and then to use this information with the Lapidus-Amundson model to predict long term movement rates. The correlation of these rates with soil and leachate properties is the basis for the predictive method which will hopefully, not require the use of a computer.

Field Verification

Limited field verification is being conducted. The initial effort[6] to date has consisted of installing monitoring wells and coring soil samples adjacent to three municipal landfill sites to identify contaminants and determine their distribution in the soil and groundwater beneath the landfill site. The sites represent varying geologic conditions, recharge rates, and age, ranging from a site closed for 15 years to a site currently operating. Individual site characteristics were identified, and sample analyses necessary to determine the primary pollutant levels in the waste soils and groundwater were determined. Validation of waste leachability and pollutant migration potential are expressed in the final report, Chemical and Physical Effects of Municipal Landfills on Underlying Soils and Groundwater, EPA-600/2-78-096, May 1978.

A second effort[5] has been recently awarded for a limited field verification of a model developed for EPA for predicting movement of landfill gases in soil. The field study will compare observed and predicted distances that methane has moved from three municipal solid waste landfills and will also compare observed and predicted efficiencies of various gas migration control systems (trenches, wells, barriers, etc.). If the model is found to provide reasonably accurate predictions, it will be used (in a later effort) to develop design charts for predicting the extent of methane migration from landfills and for developing design charts for methane control systems.

POLLUTANT CONTROL

Pollutant control studies are needed because experience and case studies have shown that some soils will not protect groundwater from contaminants. Even sites with "good soils" may have to be improved to protect against subsurface pollution. The overall objective of the pollutant control studies is to lessen the impact of pollution from waste disposal sites by technology that minimizes, contains, or eliminates pollutant release and leaching from waste residuals disposed of to the land.

The pollutant control studies are determining the ability of in-situ soils and natural soil processes to attenuate leachate contaminants as the leachate migrates through the soil from landfill sites. The studies are also determining how various synthetic and admixed materials may be utilized as liners to contain and prevent leachates from migrating from landfill sites.

Natural Soil Processes

The treatment by natural soil processes of pollutants from hazardous waste and municipal refuse disposal sites is being performed in the controlled lab studies previously discussed in the section on pollutant transport.

Moisture Infiltration Control

The initial effort[6] involves the identification of the functional requirements of cover soil and the ability of the various soil types to meet these requirements. Of primary interest is the minimization of moisture infiltration through landfill cover soils. Other criteria associated with landfill cover soils, i.e., infiltration, gas venting, vegetation, soil erosion, rodent burrowing trafficability will also be identified. The characteristics of the various soil types are reviewed individually and in combination with other soil types to determine the most suitable type of cover material for use in meeting the desired functional requirements for a given disposal site. The interim EPA report, Selection and Design of Cover for Solid Waste Disposal has been reviewed by selected groups in the user community. Field verification of the concepts described in this effort are being pursued.

Liners/Membranes/Admixtures

The liner/membrane/admixture technology[6] is being studied to evaluate suitability for eliminating or reducing leachate from landfill sites of municipal or industrial hazardous wastes. Basically, the test program will evaluate, in a landfill environment, the chemical resistance and durability of the liner materials over 12-, 24-, and 36-month exposure periods to leachates derived from industrial wastes, SO_x wastes, and municipal solid wastes. Acidic, basic, and neutral solutions will be utilized to generate industrial waste leachates.

The initial effort for liner materials being investigated under the municipal solid wastes program include six admixed materials and six flexible membranes. The admixed materials are:

- 2 asphalt concretes, varying in permeability
- 1 soil asphalt
- 2 asphalt membranes, one based on an emulsified asphalt and the other on

catalytically-blown asphalt

- 1 soil cement

The six flexible membranes are:

- Butyl rubber
- Ethylene propylene rubber (EPDM)
- Chlorinated polyethylene (CPE)
- Chlorosulfonated polyethylene (HYPALON)
- Polyethylene (PE)
- Polyvinyl chloride (PVC)

The results of the 1-year exposure have been discussed in a report entitled Evaluation of Liner Materials Exposed to Leachate-Second Interim Report EPA-600/2-76-255, Setpember 1976. The exposure of these liner materials has been extended to a 52 month period. Data on gas migration, water permeability (as in soils), osmotic, and finger print analysis is being collected for individual polymeric materials. The latest exposure results are being published in a report entitled Liner Materials Exposed to Leachate - Third Interim Report, EPA-600/2-79-038.

A second effort[6] relates primarily to the identification and description of waste disposal sites and holding ponds which have utilized an impermeable lining material. Also, three potential excavation techniques for liner recovery operations are described and discussed. This effort has been published in a report entitled Liners for Sanitary Landfills and Chemical and Hazardous Waste Disposal Sites, EPA-600/9-78-005, May 1978.

POLLUTANT TREATMENT

The pollutant treatment studies relate to the collected leachate that is physically, chemically, or biologically treated prior to discharge from the landfill site. Also, recirculation and spray irrigation concepts are considered to be potential treatment schemes. The overall objective of the pollutant treatment studies is to develop technology that treats the landfill leachate once it has been collected and contained at the landfill site.

Physical-Chemical Treatment

Various physical-chemical treatment schemes[4] were investigated in the laboratory. Physical-chemical treatments consisted of chemical precipitation, activated carbon adsorption, and reverse osmosis. The activated carbon was quite effective in removing refractory organics in the effluent of biological units. The most promising treatment scheme, an anaerobic lagoon followed by aerobic polishing, was selected for pilot plant evaluation.[1] the results of this initial effort have been reported by Ho, S., Boyle, W. C., and Ham, R. K., "Chemical Treatment of Leachates from Sanitary Landfills," JWCF, Vol. 46, No. 7, July 1, 1974, pp. 1776-1791.

A second effort[4] on the physical-chemical treatment schemes was an expansion of the initial precipitation, activated carbon, and reverse osmosis, but also in ion exchange, adsorption and chemical oxidation. The results of this effort have been reported in the Evaluation of Leachate Treatment: Volume I - Characterization of Leachate, EPA-600/2-77-186a, September 1977 or Evaluation of Leachate Treatment: Volume II - Biological Evaluation of Leachate Treatment, EPA-600/2-77-186b, November 1977.

A third effort[5] involved a laboratory evaluation of various materials that could be utilized as retardant materials to minimize migration of trace elements pollutants in landfill leachates. This investigation involves the use of landfill leachates spiked with trace elements and passed through columns of disturbed retardant materials under anaerobic conditions at controlled flow rates. The following materials have been investigated: agricultural limestone, hydrous oxides of Fe (ferrous sulfate mine waste), certain organic wastes, and soil sealants. Information to date is not sufficient to design a lining to "treat" a given amount of leachate. Though effective in retardation, hydrous iron oxides may create additional problems due to the release of reduced iron. Flux dependent pollutant retardation does not appear to be significant enough to warrant substantial work of flow rate control.

Biological Treatment

Various unit processes for biological treatment of leachate[4] have been investigated in the laboratory. The results of this initial effort have been reported in publications previously stated under "Physical-Chemical Treatment". A second effort[4] has investigated the process kinetics, the nature of the organic fraction of leachate, and the degree of treatment that may be obtainable using conventional wastewater treatment methods. The biological methods evaluated were the anaerobic filter, the aerated lagoon, and combined treatment of activated sludge and municipal sewage. Biological units were operated successfully without prior removal of the metals that were present in high concentrations. The results of this effort have been reported in the Evaluation of Leachate Treatment: Volume I - Characterization of Leachate, EPA-600/2-77-186a, September 1977 or Evaluation of Leachate Treatment: Volume II - Biological Evaluation of Leachate Treatment, EPA-600/2-77-186b, November 1977.

Recirculation

Recycling of leachate[4] is being investigated to determine the beneficial aspects of recirculation as a means of leachate control and accelerated landfill stabilization. Recommended design, operation, and control methods applicable to conventional sanitary landfill practice will be developed. This effort was discussed earlier under the "Waste Decomposition" section, and the published report is mentioned in that section. The objective of the initial effort[4] currently ongoing concerning recirculation is to confirm laboratory studies of the leachate recycle concept and to explain mass flux of gas and leachate components. Particular attention is given to the effect of evapotranspiration on the rates and quantities of leachate. An assessment of the cost/benefits of the recycle concept in accordance with daily operation/implementation including economic and technical feasibility, is being prepared.

Spray Irrigation

Spray irrigation[4] of leachate has been investigated as a low-cost, on-site treatment scheme. Optimum leachate loading rates and removal efficiencies for organic and inorganic constituents are being determined for two soil types. The technique appears to be sensitive to moisture stresses (drought).

CO-DISPOSAL

In an effort to assess the impact of co-disposal, the disposal of industrial waste materials with municipal solid waste, a project utilizing large scale experimental landfill test cells was undertaken. Concern has been voiced that the addition of industrial waste may result in the occurrence of various toxic elements in leachates and thereby pose a threat to potable groundwater supplies. Because the environmental effects from landfilling result from not only the soluble and slowly soluble materials placed in the landfill but also the products of chemical and microbiological transformations, these transformations should be a consideration in management of a landfill to the extent that they can be predicted or influenced by disposal operations.

Presently, little is known on what effect adding industrial waste has on the decomposition process and the quantity of gases and leachate produced during decomposition. There is a strong concern that addition of industrial wastes, particularly those high in heavy metals, will result in elevated metal concentrations in the leachates and potentially, in potable groundwater supplies. Advocates of co-disposal of sludges and municipal waste believe the presence of organics in the landfill will immobilize heavy metals. They also believe the presence of such sludges may accelerate the decomposition process and shorten the time required for biological stabilization of the refuse. Because of the high moisture content and, commonly, the high pH and alkalinity of these sludges, periodic analyses of the leachates in this study for trace and heavy metals is expected to provide data to allow rational evaluation of the practice of co-disposal. The over-activity is to evaluate and develop a predictable formulation for the transformation process.

The initial effort[4] involves a study of the factors influencing (1) the rate of decomposition of solid waste in a sanitary landfill, (2) the quantity and quality of gas and leachate produced during decomposition, and (3) the effect of admixing industrial sludges and sewage sludge with municipal refuse. A combination of municipal solid wate and various solid and semi-solid industrial wastes was added to several field lysimeters. All material flows were measured and characterized for the continuing study and related to leachate quality and quantity, gas production, and microbial activity.

The industrial wastes investigated were: petroleum sludge, battery production waste, electroplating waste, inorganic pigment sludge, chlorine production brine sludge, and a solvent-based paint sludge. Also, municipal digested primary sewage sludge dewatered to approximately 20 percent solids was utilized at three different ratios. The results of this initial effort have been reported by Streng, D. R., "The Effects of Industrial Sludges on Landfill Leachate and Gas," Proceedings - National Conference on Disposal of Residues on Land, September 1976, pp. 69-76. The updated results of this effort are being published in a report entitled Pilot Scale Evaluation of Sanitary Landfilled Municipal and Industrial Wastes.

A second effort[5] to assess the potential effects of co-disposal involves the leaching of industrial wastes with municipal landfill leachate as well as water. Leachate from a municipal solid waste (MSW) landfill was used to extract the five industrial wastes and to study movement of their components in the soil columns. MSW leachate dissolved much greater amounts of substances from the wastes and apparently increased the mobility of these substances in the soil columns relative to the dissolution and mobility observed when deionized water was used as a leaching solution. The municipal landfill leachate is a highly odorous material containing many organic acids and is strongly buffered at a pH of about 5. Consequently, it has proved to be a very effective solvent. A sequential batch leaching and soil adsorption procedure has been developed that provides information comparable to that from soil column studies but in a much shorter time.

A third effort[6] involves a study of the effects of co-disposing of chemically stabilized sludges in a municipal refuse landfill. The stability, weatherability, pollutant leachability, and leaching rate of raw and fixed sludges will be determined. Test results show that fixing can cause significant changes in the properties of sludge, that fixed sludges are similar to soil, soil-cement, or low strength concrete, and that properties are process dependent.

ENVIRONMENTAL ASSESSMENT

The environmental effects of waste disposal to the land need to be determined in relation to the management and disposal practices for municipal solid wastes. In an effort to assess the impact of these practices, several studies have been initiated. The overall objective is to develop predictive procedures for forecasting adverse environmental effects from land disposal activities and to provide user documents for implementing field practices for those methods that eliminate or minimize adverse effects.

The initial effort[7] involves determining the effects of application of composted municipal wastes and sewage sludge on selected soils and plants of croplands. Multiple applications of composted municipal refuse totaling 900 metric tons per hectare have resulted in satisfactory crop growth with only a moderate increase of some heavy metals in plant tissues. Very little downward movement of heavy metals was observed under conditions of heavy leaching in greenhouse or natural outdoor conditions. The effort has been published in a USEPA report entitled Effect of Land Disposal Applications of Municipal Wastes on Crop Fields and Heavy Metal Uptake, EPA-600/2-77-014, April 1977.

A second effort[6] involves a determination or evaluation of vegetation kills and growth problems associated with landfill gas migration as evaluated by mail survey and on-site investigations. Additional investigations are being performed to determine control measures for reducing vegetation losses, and experimental plot observations should determine those vegetation species most conducive to landfill environs. The results of this initial effort have been reported in Study of Vegetative Problems Associated with Refuse Landfills, EPA-600/2-78-094, May 1978.

A third effort[6] in this area involves the screening of woody species for adaptability to landfill conditions of assessment of planting techniques for exclusion of landfill gases from root zones. Trees found to be unsuitable for landfill conditions will be replaced with other potentially adaptable species to obtain additional tolerance information. The study will also assess the value of mycorrhizal fungi in reducing tolerance of trees to landfill conditions.

A fourth effort[1] has been initiated to study the operational and asthetic effects of milled refuse particle size in a landfill operated without daily cover. The overall objective of this research is to establish acceptable parameters for the operation and maintenance of milled refuse landfills in order to minimize detrimental environmental effects. Specific variables to be evaluated are: the effect of wind velocities and direction on the movement of landfilling material; the amount of differential settlement associated with particle size variations; the initial density in each test cell and subsequent density with relation to time and consolidation within the cell and the presence or absence of surface crusting; qualitative evaluation of nuisance organisms, wildlife, and the type and amount of plant growth; and some evaluation of odors and background conditions potentially responsible for noticeable odors.

A fifth effort[2] compares the impact on ground and surface waters of alternate types of landfills: millfill, balefill, hillfill, strip mine fill, and permitted sanitary landfill. The effort will include studies of groundwater, surface waters, in-situ soil conditions, hydrogeology, and water mass balance at each site.

REMEDIAL ACTION

An ongoing study by OSW has identified incidents of well contamination due to waste disposal sites. Seventy-five,to 85 percent of all sites investigated are contaminating ground or surface waters. In order to determine the best practical technology and economical corrective measures to remedy these landfill leachate and gas pollution problems, a research effort has been initiated[2] to provide local municipalities and users with the data necessary to make sound judgments on the selection of viable, in-situ, remedial procedures and to give them an indication of the cost that would be associated with such a project. This research effort is being pursued at a municipal refuse landfill site at Windham, Connecticut. The site was selected from a candidate site list of 17 which took into account appropriate site selection criteria. This effort consists of three phases. Phase I will be an engineering feasibility study that will determine on a site specific basis the best practicable technology to be applied from existing neutralization or confinement techniques. Phase II will determine the effectiveness, by actual field verification, of the recommendations/first phase study. Phase III will provide a site remedial guide to local municipalities and users. The engineering feasibility study has been completed and a report entitled Guidance Manual for Minimizing Pollution from Waste Disposal Sites, EPA-600/2-78-142, August 1978 has been published. This guidance document emphasizes remedial schemes or techniques for pollution containment. The remedial schemes discussed in this document are:

- Surface Water Control
  - o surface sealing
  - o revegetation
- Groundwater Control
  - o bentonite slurry-trench cutoff wall
  - o grout curtain
  - o sheet piling cutoff wall
  - o bottom sealing
- Plume Management
  - o extraction
  - o injection
  - o leachate handling
- Chemical Immobilization
  - o chemical fixation
  - o chemical injection
- Excavation and Reburial

The scheme currently being considered for the Connecticut MSW site is a surface sealing technique which could be followed by a leachate extraction scheme if required.

A second effort[2] involved the sponsorship of a conference of nationally recognized authorities in the area of Solid Waste Disposal to discuss alternative remedial procedures that could be implemented at the Army Creek Landfill in New Castle County, Delaware. This three day "Round Table Discussion on Army Creek" was held on November 16-18, 1977. It is planned that discussion and the consensus of this meeting is to be published in a proceedings document.

LANDFILL ALTERNATIVES/LAND CULTIVATION

Municipal solid wastes are primarily deposited in standard sanitary landfills or incinerated. Because of concern for environmental impact and economics, other landfill alternatives have been proposed. For SHWRD purposes, the alternatives currently being considered are: (1) deep well injection, (2) underground mines, (3) land cultivation, and (4) saline/marshland environments. The deep well injection and underground mine alternatives are strictly orientated towards hazardous wastes and as such will not be discussed in this symposium or its proceedings. The remaining two alternatives are, however, orientated towards municipal waste. In order to assess the feasibility and beneficial aspects of spreading and admixing municipal refuse into the soil, an initial effort[6] involves land cultivation and the preparation of a state-of-the-art document. Available data indicate that application of shredded municipal refuse or compost to marginal or drastically disturbed land improves soil structure and fertility, thus making revegetation possible. It appears that the environmental pollution caused by land cultivation is minimal as compared to that for landfills, primarily due to maintenance of aerobic conditions and the lower concentration of waste per unit area of land. Technical and economic assessment efforts will follow. Results of this study were presented in reports entitled, Land Cultivation of Industrial Wastes and Municipal Solid Wastes: State-of-the-Art Study, Volume I - Technical Summary and Literature Survey, EPA-600/12-78-140a, August 1978 Land Cultivation of Industrial and Municipal Solid Wastes: State-of-the-Art Study, Volume II - Field Investigation and Case Studies, EPA-600/2-78-140b, August 1978.

A second effort[1] involves documenting the disposal of municipal solid wastes in saline/marshlands. The purpose of this study is to obtain a document detailing the present environmental and economic status of municipal solid waste disposal into specific saline environments and compiling state regulations and policies in effect for those states bordering saline waters. Information was obtained by a questionnaire and over 141 instances of MSW disposal in coastal saline environments have been identified in 17 states. Seventy-seven (or more) have monitoring data available.

ECONOMIC ANALYSIS

An initial effort[8] is being planned to evaluate the relative importance of the various factors affecting the cost of sanitary landfilling of solid waste and to provide a method for evaluating the cost of combinations of waste processing and sanitary landfilling. This will identify the least cost disposal alternative under local conditions. It is intended to provide useful information to local decision-makers and municipal solid waste managers in planning and operating their sanitary landfill operations. The need for this information is especially critical in view of the guidelines to be developed under Subtitle A of the RCRA of 1976.

CONCLUSION

The laboratory and field research project efforts discussed here reflect the overall SHWRD effort in municipal solid waste disposal research. The projects will be discussed in detail in the following papers. More information about a specific project or study can be obtained by contacting the project officer referenced in the text. Inquiries can also be directed to the Director, Solid and Hazardous Waste Research Division, Municipal Environmental Research Laboratory, U.S. Environmental Protection Agency, 26 West St. Clair Street, Cincinnati, Ohio 45268. Information will be provided with the understanding that it is from research in progress and that conclusions may change as techniques are improved and more complete data become available.

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SOLID WASTE DISPOSAL RESEARCH ACTIVITIES OF THE FEDERAL GOVERNMENT IN CANADA

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ABSTRACT

In Canada the Federal government's role is principally to develop solid waste management technology, to generate and evaluate technical solid waste management information, and to develop new or improved guidelines or codes of good practice relating to various waste management aspects. These activities are conducted on a national basis for the benefit of the industry and regulatory agencies.

Solid waste research activities in Canada are being undertaken primarily at the federal government level. Current research activities, relating to land disposal in particular, are underway in each of the following study areas: landfill leachate research; municipal waste disposal site selection; landfill field investigations: co-disposal studies; landfill gas research; and the NATO/CCMS landfill project. This paper describes Environment Canada's current research projects in these areas of interest.

INTRODUCTION

Solid waste research activities in Canada are being undertaken mainly at the federal level of government, but also at the provincial government level, in order to come to grips with the problems facing both the private sector and the regulatory agencies. Some provincial governments are actively contracting out basic research studies or undertaking similar in-house investigations, although most others are simply studying localized problems of disposal or local potentials for resource and evergy conservation. The bulk of the research effort in Canada is undertaken by the federal Departmentof the Environment (Environment Canada).

The Waste Management Branch of Environment Canada is very active in the resource conservation and the municipal and industrial waste management areas of responsibility. However, the constraints of this presentation to be the activities of our Branch relating to the disposal of wastes onto land.

These activities will be briefly described under the following headings:

1. landfill leachate research;

2. municipal waste disposal site selection;
3. landfill field investigations;
4. co-disposal studies;
5. landfill gas research; and,
6. the NATO/CCMS landfill project.

LANDFILL LEACHATE RESEARCH

Leachate Production

Although a significant amount of research work has been accomplished to date on the subject of leachate generation, or production, at municipal solid waste landfills, there are still many basic questions left unanswered about the actual process of leaching. Leachate production is generally an estimated variable, and never really measured directly at a landfill site.

The ability to predict leachate quantity and quality is probably the most necessary feature required for the design of leachate control systems, including natural attenuation and leachate treatment systems. All landfill leachate control systems design work relies heavily on an accurate input function describing the leaching behavior of the landfill. An estimate is only as good as the "rule-of-thumb" used, and therefore, we have initiated an attempt to develop a simple waste leaching model at the University of British Columbia in Vancouver.

A good quantative, descriptive and predictive model of the leaching of solid waste landfills and of the mechanisms associated with the leaching process may be important to assess the environmental impact of leachate on receiving waters; to aid in the design of any required leachate treatment facilities; to provide estimates of the material inputs into an underlying groundwater system as a first step in determining leachate concentration there: and, to assess the likely effects of co-disposal of various liquid and/or semi-solid wastes with municipal solid waste in landfills. For the initial stages of this work a model was chosen which assumes a simple physical process to describe the leaching. The initial model assumes flow-through of infiltrating water and a mass transfer of soluble materials from the solid phase to the liquid phase due to a concentration difference driving force, as well as connection within the liquid phase. The model has been tested for the simple case of plug flow starting constant rate or infiltration, insignificant moisture in the refuse, and a uniform distribution of leachable materials.

This model provided leachate concentrations as a function of time. The calculations require values for the refuse depth, the rate of infiltration, refuse field capacity, and three empirically determined parameters. The model has been fitted to data from solid waste lysimeter experiments for six contaminant curves and a range of operating conditions. The calculated curves and corresponding data curves generally showed a good correlation when the empirical parameters were suitably adjusted.

The model is currently being further modified to provide both the quantity and quality of leachate as a function of time, and for various patterns of infiltration and fluid flow. Other variables being tested are various landfill operational parameters. A final report on the initial work should be available within the next few months. Leachate Control and Treatment Infiltration into disposed refuse results in the production of leachate which is recognized to be potentially hazardous to the environment. There are a significant number of documented case histories which show that the uncontrolled migration of leachate from improperly located waste disposal sites can result in extensive groundwater and/or surface water pollution. These case histories have been used to demonstrate the need for the design of leachate control and treatment systems at waste disposal sites.

There is a growing trend in regulatory agencies insisting on some degree of leachate control including collection and treatment. This trend is disconcerting in that many leachate control systems have not been proven technically, while others have simply been designed to expedite the approvals process. Furthermore, some systems may be failing because of inadequate design or improper installation, whereas others are being over-designed without due regard to the cost-effectiveness of the systems. Rarely has the environmental benefit and the complete cost and benefit of such a system been evaluated.

Landfill designers, researchers, and regulatory agencies have become concerned with problems relating to proper selection, design, and operation of leachate control and treatment systems. There is a recognized need to develop practical criteria for the design and operation of such leachate management systems.

In response to this need, the Waste Management Branch and its contractors sponsored two international round-table discussions on the topics of leachate control, held in Vancouver, and leachate treatment, held in Halifax.

The session on leachate control considered the topics of natural and engineered controls to minimize leachate production, collect leachate on-site, recirculate leachate, and rectify leachate migration problems through aquifer manipulation as a control procedure. A report is being prepared on the recommended procedures for leachate control based on a concensus by technical experts from the U.S. and Canada.

The purpose of the leachate treatment session was to discuss and document current treatment concepts, practices, and methodologies.

Attendees at this session included recognized treatment experts from Canada, the U. S., Germany and Norway. A report is being prepared to document the discussion on the need for and the factors affecting leachate treatment; the treatment of leachate in the natural hydrogeologic environment; biological treatment systems physical-chemical treatment systems; and, the concensus of expert opinion expressed at the session.

With regards to in-situ attenuation, it was recognized that there is a difficulty in designing for the use of the natural setting as an attenuation and/or treatment system, and although it was agreed that the assimilative capacity of the hydrogeologic environment for leachate contaminants is quite significant, ground water contamination will usually result.

The concern, therefore, must be with the degree of contamination which will be allowed, and whether such contamination actually constitutes pollution.

It was recognized that all constructed treatment systems require, prior to design, the setting of an effluent quality standard. Similarly, if the hydrogeologic environment is to be designed to accept a leachate loading, water quality standards at a finite distance downgradient from the base of the disposal area must be established. Most commonly, the maintenance of drinking water standards has been used as a criterion for leachate discharge in the hydrogeologic environment.

A realistic leachate discharge objective would be that leachate must not impair a reasonable, legitimate, present or potential off-site groundwater use. The principal problem which makes it difficult to define an effluent standard for the discharge of leachate to the hydrogeologic environment, is that in general, the hydrogeologic environment cannot be improved to allow for an upgrading of the standard if necessary. At the treatment facility, such an upgrading can readily be facilitated.

It could be argued that all landfill designs, where leachate is discharged to the hydrogeologic environment, should be equipped with contingency plans which are designed to control the discharge of leachate to this environment should it be required. If a contingency plan can successfully be implemented, a setting of design standards could be readily done.

With respect to engineered treatment systems, it was agreed that current processes can produce effluent liquids virtually free of contaminants, notwithstanding the high costs involved. However, at the same time, rather than being destroyed or converted to a less objectionable form, the contaminants are either retained in a highly concentrated residual stream or transferred to another medium, both of which require disposal.

Most treatment processes have as their objective, the production of a high quality effluent liquid without regard for the residual stream. Since landfills are frequently the last step in disposal, simply transferring leachate contaminants from one form to another for disposal "elsewhere" was not considered to be a satisfactory solution to a leachate problem.

Many solid waste disposal systems are not presently in a position to finance "best available technology".

Consequently, it is hoped that, for cases where leachate treatment is clearly deemed necessary, it will be done according to the "most practicable technology" after consideration of all relevant factors has been completed. This should be based on realistic assessment of the potential of, and expectations for, the disposal area, be it a surface or a groundwater system, and the resources at hand to solve the problem.

Reports on both the control and treatment sessions will be available soon.

Extensive research into leachate treatment has been underway for the last few years at the University of British Columbia, and an in-depth reporting of this program will be presented later in this symposium by Dr. Don Mavinic of their Civil Engineering Department.

Leachate Toxicity

At the third annual research symposium in St. Louis, I briefly mentioned our work in developing a rapid leachate toxicity testing procedure. The procedure, using rainbow trout fingerlings in stoppered BOD bottles, proved satisfactory for many test situations.

Toxicity has often been suggested as a valuable indicator parameter in surface waters monitoring programs. However, standard bioassay procedures are expensive and generally require large volumes of effluent, and may take up to three days per sample The rapid toxicity procedure was developed to overcome these problems, and it was demonstrated to be a less expensive, valuable in-situ monitoring procedure.

A final report on the set-up and use of the procedure has been pubished.

Our continuing interest in leachate toxicity led us to develop an even simpler test using Daphnia, and an attempt to attribute leachate toxicity to concentrations of specific leachate contaminants. In addition, work was done to determine p^H effects on toxicity, and to assess the effects of landfill operating variables such as rainfall, sludge additions, and elapsed time since disposal, on leachate toxicity.

Equations were developed by statistical regression analyses of the experimental data to relate toxicity to contaminant concentrations. It was found that the relationship between toxicity and contaminant concentrations can best be expressed in terms of the un-ionized ammonia concentrations as well as the concentration of zinc in the case with the Daphnia test, or the copper concentrations in the case with fish tests.

Furthermore, it was reported that the adjustment of p^H in test procedures can dramatically reduce the toxicity of a given sample.

A final report is in preparation.

MUNICIPAL WASTE DISPOSAL SITE SELECTION

Accepting that landfilling will be a requirement for many years to come, it is necessary that technology required to locate and manage these sites be developed to a high degree.

Over the past year we have been continuing our development of a selection procedure for locating suitable landfill sites which could be used to service small communities. The procedure will be a simple, yet comprehensive attempt to provide a small community with a decision making process which will assist them in locating the most suitable site from an environmental, a public protection, and an operation and management point of view, without having to rely on outside expertise. A "cook book" approach is being used in preparing a final report on the procedure.

The selection procedure works as follows. Potential sites are evaluated against environmental protection criteria, including distance requirements to property boundaries based on acceptable attenuation distances within the local soil environment, distance requirements to downgradient water wells, and distance requirements to ensure protection of nearby dwellings from landfill gas migration problems. Other criteria also apply.

A pass-fail test is used for each criteria. Failure to pass any of the criteria would render the site unsuitable without special site engineering.

From a public protection point of view, the major criteria of importance in addition to environmental and health criteria are aesthetics, traffic, land use designations and site completion. The operation and management criteria ensure the selection of the most operable and cost-effective location.

The procedure is based partially on a variety of procedures documented to date, and should prove to be a simple, yet effective decision tool for decision makers in small communities.

The procedure should be finally available within the next few months.

LANDFILL FIELD INVESTIGATIONS

Although a great deal of research has already been devoted to landfill technology the long and inconclusive arguments between experts over the acceptability of particular sites clearly points to the need for additional basic technical information. These arguments are most often prevalent at public meetings or regulatory hearings where frequently neither side of the argument can be adequately supported by firm technical information.

In spite of the large amount of research which has already been conducted, problems arise as a result of the difficulty in showing the applicability of laboratory data to field conditions and in the difficulty in studying the processes directly in the heterogeneous hydrogeologic environments of the real world.

Because a large amount of detailed information has already been gathered at the C.F.B. Borden landfill, as reported in previous years, this site offered us an excellent opportunity for further investigating both the applicability of laboratory data for making field predictions and for the field investigation of contaminant migration and attenuation processes.

Several activities have been undertaken, and will be continuing, at this landfill site. One such activity includes the further definition of the physical hydrogeology by supplementing information from our past studies with some continuous core sample analysis for detailed stratigraphy, point dilution testing for direct velocity measurements, single well response tests for determining in-situ K values, and field dispersivity measurements.

Other activities at the site have included the geochemical surveying of the contaminant plume through field measurements of Eh, p^H, DO and specific conductance taken on samples extracted from multilevel sampling devices.

Trace metal-oxide coating studies are also being undertaken to determine the contribution to contaminant levels in the downgradient soils due to the release of trace metals from soil oxide coatings being dissolved under reducing conditions. Studies have shown that the bulk of the trace element contect of contaminated ground water may be derived not from the waste material itself, but from the natural manganese and ferric oxide coatings on the surrounding material through which the leachate moves. Another part of the field studies consists of the evaluation of the degree on anaerobic decomposition of leachate organics migrating through saturated soils under anaerobic conditions. Initial results have shown a COD reduction of 86% after 1.5 meters of travel and a TOC reduction over the same distance of 90%.

A comprehensive modeling exercise completes the long list of activities being undertaken at the Borden landfill. The development of a calibrated flow model is being followed by a simulation of the distribution of a non-reactive solute, namely chloride, and then a reactive species. Lastly, the effects of transient flow conditions will be examined .

One important aspect of this modeling exercise is to determine a suitable input function for the contaminant release from the landfill itself. In the absence of historical data from the site, a successful leaching model such as the one being developed at the University of British Columbia may provide necessary information.

If modeling is to have a place at all in our future work, such as has been suggested by Weston in their report to EPA on "Pollution Prediction Techniques for Waste Disposal Siting", then the Borden work should be the "proof of the pudding".

CO-DISPOSAL STUDIES

Septic Tank Sludge and Municipal Refuse

Our previous research project, on the effect of adding septic tank pumpings to refuse has been completed. The general finding of this research was that the addition of septic tank pumpings, in sufficient quantity, significantly reduced the mass of contaminants discharged, and particularly heavy metals, when compared with a control containing no pumpings. This effect, attributed largely to biological activity, was reduced when rainfall rates were increased from 15 in (381 mm) to 45 in (1143 mm) per year. Increased gas production and a greater initial percentage of methane gas combined with reduced contaminant mass discharges at the high septic tank pumping loading rates made this technique look promising from both an environmental and an energy recovery viewpoint. Before proceeding with a large scale operation, however, it was felt necessary to determine whether or not the tie up of metals was permanent or simply transient in nature.

It was decided, therefore, to continue to apply water to the existing 9 lysimeters at the rates which had been previously used, and to analyze gas and leachates on a continuing monthly basis.

Rainfall applications and leachate collection have been continued since the termination of the original contract, and with only three exceptions, all leachate contaminant concentrations have continued to decrease. The exceptions are one tank which showed increase in colour but no other increases; another tank showed an anamalous steady increase in iron concentration; and yet another showed an initial increase in iron concentration but then a drop to the previous low level. Generally, p^H HAS INCREASED. Concentrations or metals such as cadmium and lead decreased to below detection limits for most tanks.

There is presently no indication that desorption of contaminants is occurring. However, a continued increase in gas production in some tanks shows that biological activity is continuing, and if biological activity is the mechanism which is causing metal tie-ups, this indicates that desorption may not occur for some time yet, if at all.

A final report should be available this summer, 1979.

Metal Sludges and Municipal Refuse

Having demonstrated the beneficial effects of adding untreated biological sludge to landfills in terms of achieving metal tie-ups, we decided to undertake further work to determine whether or not these previously found beneficial effects would apply when industrial sludges were added to the refuse as well.

A series of thirteen lysimeters were constructed using 12 inch diameter PVC pipe. Flat bottoms were welded to each. A specially designed sealed tops were constructed. The necessary additions of septic tank sludge and an arsenic containing waste and an electro-plating sludge were each determined based on past codisposal experience.

The tanks were filled and put into operation in August 1978. Results to date are still being analyzed, and the study will continue into 1981.

LANDFILL GAS RESEARCH

Gas Production and Migration

The objective of our gas production and migration study is to provide information about gas composition, pressure, and production in landfills. Such information is essential for both an analysis of gas migration into adjacent soils and an assessment of gas recovery potentials at landfill sites. This study, to be discussed in detail later in this program by Dr. Grahame Farquhar of the University of Waterloo, will also involve the use of field data in the calibration of a previously developed gas transport model. The calibrated model will be used to display variations in gas migration patterns and to evaluate various gas control and recovery schemes.

Gas Recovery and Utilization

Although it is being recognized that a municipal solid waste landfill may be a significant energy source, it is clearly not feasible to tap smaller sites in remote locations for low BTU content gas supplies for consumption elsewhere. Unfortunately, there is usually no nearby gas distribution system available. Even if a pipeline could be found next to every small site, the economics of cleaning the gas to pipeline quality could be prohibitive. Our study addresses the possibility of extracting landfill gases year round at any site and using these gases for an on-site energy supply. One market which could readily benefit in Canada from such a readily available supply is the energy intensive greenhouse industry, which presently faces fuel costs of up to $30,000 per year per greenhouse acre.

California studies have suggested that the best economics lie in the direct use of processing and refining. Our 10 month study is intended to demonstrate the use of the collected gas to supply the energy requirements of a small greenhouse throughout the year. It is planned to grow tomatoes in this greenhouse situated on a completed site in St. Thomas, Ontario.

The completed St. Thomas Landfill Site is divided into two sections. The section under study was in operation from 1967 to 1978. This section is comprised of approximately 24 acres of refuse with an average depth of 40 feet. The gas production wells, the gas monitoring and pressure probes, and the various greenhouse facilities.

The gas production well consists of a 36 inch augered bore hole, logged as 1 foot of fill, 28 feet of refuse and 3 feet of native soil. A 6 inch diameter PVC pipe with an attached 8 feet of stainless steel well screen (50 slot) was placed in the centre of the augered hole. The hole was then backfilled with pea gravel, with gas pressure and monitoring probes inserted at various depths. The top of the gas production well was closed using 1 1/2 feet of 36 inch diameter concrete tile. Bentonite was placed around the tile and a concrete plug was used at the surface. A concrete cap is placed on top of the concrete tile.

A rubber adapter is being used to connect the pipe leading from the gas pump to the concrete cap, so as to allow pumping directly off the gravel pack to permit an assessment of the feasibility of pumping off such an inexpensive gravel pack without having to install gas extraction well hardware. After completion of pumping from the gravel pack, 6 inch PVC pipe will be connected directly to the gas line feeding from the gas pump.

The gas recovery system extracts the gas from the production well, passes the gas through a water knock-out drum and either pumps the gas to a furnace within the greenhouse or exhausts the gas to the atmosphere.

The control on the gas system will be a pressure release valve. The pump will run continuously at a set flow rate, and if the pressure goes up in the line, the pressure release valve will allow gases to exit to the air. When the furnace is not in operation, all the gas will be released to the air.

A forced air furnace has been adapted to utilize methane gas. A back-up electric furnace will be installed in case of system failure, to prevent crop damage due to frost.

A total of 46 gas monitoring and pressure probes were installed. In addition, 3 gas monitoring and pressure probes were installed below the greenhouse liner to monitor for gases coming from the fill underneath the greenhouse.

The complete system is currently in operation, and methane concentration and pressure distributions within the landfill, changing as a result of varying pumping rates, are being monitored.

The study will be completed in August, 1979 and a final report should be available shortly thereafter.

NATO/CCMS LANDFILL PROJECT

A major international effort is underway among the member countries of NATO, including the U. S. and Canada. The purpose of the pilot project is to study hazardous wastes management, and one part of this project is a landfill study being supported by the U. S., Canada, Norway, the U. K., the Netherlands, and the Federal Republic of Germany. The purpose of the landfilling study is to assess the role of landfilling in hazardous waste disposal. As Chairman of this sub-project for the next three years, I will be reporting on our deliberations and coordinating input from the various landfill research programs being undertaken by each participating country. All submissions will be used to prepare a final project report to NATO.

The results of our work should have a significant impact on policy planning within the member countries.

One of the topics to be addressed will be the co-disposal of industrial and municipal refuse. Member countries at present appear to have opposing view points on the merits of co-disposal, and the resolution of this issue, if at all possible on the basis of research results, should prove to be a most interesting challenge.

Another very important aspect of our joint international undertaking is the expressed desire by some member countries to pursue future research efforts on a cooperative basis and to establish a formal information exchange.

In accordance with this spirit of developing cooperative international efforts, we have already initiated a coordination with the Solid and Hazardous Waste Research Division of EPA of our respective landfill gas research studies, in the hope that we can mutually benefit from each other's work from the outset, and avoid costly duplication. It may even be possible to directly tie into similar German studies now getting underway. The benefits of such an international coordination of efforts will obviously accrue to us all.

The landfill sub-project will be completed in late 1980.

EFFECT OF MOISTURE REGIMES AND TEMPERATURE ON MSW STABILIZATION

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ABSTRACT

The production rate and the composition of gases released during anaerobic degradation of solid waste in laboratory-simulated landfills were measured. The major gases observed were CO_2, H_2 and CH_4. The moisture content, which was found to be the most important variable affecting the rate of gas production, must be above 75%, based on the dry weight of solid wastes, but below 100%, or field capacity, to maximize gas production while holding the generation of leachate to an acceptable minimum. The initiation of methane generation is strongly influenced by the pH and buffer conditions of the solid waste moisture.

INTRODUCTION

Landfills are considered to be the ultimate sites for disposal of solid wastes. A stabilized, well-engineered and well managed landfill, such as an approved sanitary landfill, is usually much more environmentally acceptable than a poorly designed, inappropriately operated and unstabilized one. In fact, under the new solid waste law (Pl-94-580), wastes, residues and hazardous materials cannot be disposed of in other than approved sanitary landfills. After the solid wastes are placed in the sanitary landfills, some impairment of environmental quality, however, can result from the release of explosive gases produced by decomposing refuse and the movement of leachate generated by rainwater infiltrating the solid wastes. One way to minimize such insult to the environment is to accelerate the process of stabilization of solid wastes and to recovery by-product of value, e.g., the methane gas.

A number of studies has concluded that it is technically feasible to recover methane from anaerobically decomposing solid waste in landfills (1-6). Augenstein et al. (7) have shown improvement of converting shredded solid wastes to fuel gas by employing finely divided calcium carbonate as a pH buffer. The rate and amount of gas produced, however, are uncertain, as is the influence of environmental factors such as moisture content, density, waste particle size, temperature, exposure to air and buffer conditions. Because systematic testing is best accomplished on a laboratory scale, the bench-scale study described here was initiated to evaluate the effects of these factors. Shredded municipal solid waste was placed in sealed containers, and the production and composition of the generated gases were measured. The characteristics of generated leachates were also determined during the transient-state of adding simulated rainfall.

MATERIALS AND METHODS

Eighteen 208-liter (55-gallon) steel containers of the type shown in Figure 1 were used in the laboratory simulation. Each drum was lined with 10 polyethylene bags of 0.15-mm thickness. A cushion of construction sand was placed between the drum floor and the plastic liner, with a gradual slope to a height of about 8 cm at the periphery of the drum to funnel leachate towards a central drain. A 15-cm layer of class A gravel was placed inside the liner to provide for collection of the leachate and screening of the drain. Two fittings were installed on the top of each drum. One was used to apply water to the solid waste and the second allowed for the collection of gases and provided access for the thermocouple wires. The drain installed at the bottom of the container was used to collect the leachate.

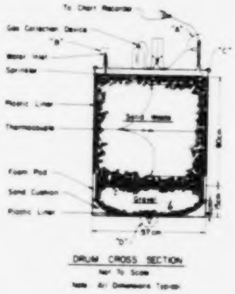

Figure 1. Cross-Section of Solid Waste Container

The containers were filled with from 55.0 to 80.5 kg (dry weight) of shredded solid waste. They were then sealed and maintained under different environmental conditions. Seventeen of the containers were located in an insulated and air-conditioned room maintained at an average temperature of 17 C (62 F). One was located in an adjacent room maintained at an average temperature of 26 C (79 F).

The solid waste was obtained from the City Solid Waste Reduction Plant in Madison, Wisconsin. It had been collected by municipal employees in wards 6, 7, 13, and 15 on Thursday, January 15, 1976. In ward 6, located in downtown Madison, the values of the homes were generally less than $10,000. Ward 7, also located in downtown Madison, is composed largely of old homes in the $20,000 range, many of which are occupied by students. Wards 13 and 19 are both in suburban areas with homes in the $50,000 and $30,000 ranges, respectively. The solid waste is collected once a week and is brought to the reduction plant where it is milled by a Tollemache mill and shredded to a nominal size of 0.7 to 2.5 cm.

The gas produced in the test cells was collected with Mariotte flasks consisting of two bottles, one of which was placed at a higher level than the other and filled with water (Figure 1). As the gas was produced, it flowed into the upper bottle, displacing water into the lower bottle. The volume of gas produced was determined by measuring the change in water levels. Gases were collected intermittently rather than continuously, and the drums were kept completely sealed except for brief periods during gas measurement.

Simulated rainfall was being applied after the initial phase of studies on steady-state moisture content. During the transient-state testing, a buffering substance, $NaHCO_3$, at a concentration of 25.5 g/ℓ was added to one of the test cells to observe its effect on both the rate of gas production and the stimulation of methane formation.

RESULTS AND DISCUSSION

The initial data collected in the present study have been presented previously by Chian *et al.* (8) and DeWalle *et al.* (9). All containers produced gas, the amount varying between 0.09 and 6.0 ℓ/kg dry weight, as shown in Figure 2. Details of the experimental conditions were given in the aforementioned articles (8,9).

The effect of the different environmental variables can be deduced from the data in Figure 2. The largest amount of gas was produced in containers 6, 13, and 14, all of which were maintained at the highest moisture content, i.e., 99%, based on the dry weight of solid wastes. Substantially lower quantities were produced at 60% and 36% moisture. The size of the solid waste is probably the second most important factor, as shown by the gas production from cells 4, 15, and 18. The moisture content of these three cells was maintained at 78%, but the cells contained waste of different sizes. Cell 18 contained 12.5-cm solid waste, and cell 4 contained 25-cm waste, whereas all other cells (including cell 15) contained 2.5-cm waste. Increasing the density of the solid waste tends to decrease the gas production (cell 12 and 16, Figure 2), possibly because of a decrease in the effective surface area exposed to enzymatic hydrolysis. Temperature is shown to be the environmental variable having the least effect on gas production (cell 1, 9, 7 and 11, Figure 2).

While the gas production data for the containers clearly showed the effects of the environmental factors, significant variability was found among individual containers maintained under identical conditions. The highest gas-production rate at the 36% moisture content, for example was 265% of the average value, while the lowest rate was 14% of the average. At higher moisture contents the variability was less; the highest rate at 99% moisture was 113% of the average, and the lowest rate was 88% of the average. The small variation in the rates at 99% moisture is probably a result of the homogeneous distribution of moisture within the drums.

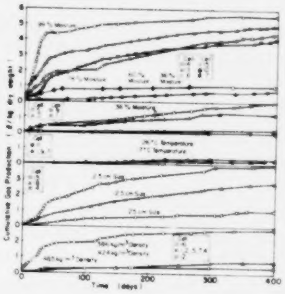

Figure 2. Cumulative Gas Production of the Enclosed Solid as Affected by Different Variables During Steady-State Testing

The results in Figure 2 further indicate that in cells having high gas-production rates, the gas is produced first at a high rate and then at a reduced rate which is a factor of 20 to 40 less than the initial rate. The high initial rates, which were sustained for up to 40 days, indicate that a major amount of the gas-producing solid waste fraction is readily and rapidly degradable, probably at zero-order rates. The shape of the other cumulative gas production curves generally do not show this stepwise pattern. Instead, they resemble more closely a parabolic function, possibly reflecting first-order reaction rates.

Gas analysis showed that the N_2 initially present at 79% was rapidly replaced by CO_2. The most rapid N_2 displacement was observed in the containers with the highest moisture contents, reflecting the CO_2 "bloom" often observed in landfills (10). After approximately 50 days, hydrogen was found in the gas phase from 8 of the 15 sealed containers (9). The amount of hydrogen peaked after approximately 100 days and gradually decreased thereafter. Only one of the containers filled with large waste particles (cell 4) has produced methane during the initial phase study with constant moisture content. The methane, which constituted 50% of the gas volume, appeared immediately in the container rather than following the typical sequence in which methane generation follows the CO_2 bloom (10). The gas composition data therefore indicate that at the time of the most recent measurement most of the containers had just completed the acid hydrolysis phase characterized by CO_2 and H_2 generation. Methane fermentation apparently did not start in most containers within the initial one-year period.

As the moisture content was found to be the most important variable during the initial 400-day monitoring period, this parameter was studied in greater detail during the following 500-day period. Six of the 15 containers received simulated rainfall at a rate of 25 to 50 cm/year to simulate transient-state conditions, as opposed to the steady-state conditions maintained during the first period. The resulting data are plotted in Figure 3, with day 0 representing the end of the first 400-day period and the beginning of the transient-state testing.

The results presented in Figure 3 show that the greatest gas production occurred in the containers that were brought to a 99% moisture content at the beginning of the first period (cells 6, 13 and 14). Addition of 25 cm/year of simulated rainfall to cells 13 and 14 immediately produced leachate, as the 99% moisture content was approaching field capacity. Adding a buffering substance, 25.5 g/ℓ of $NaHCO_3$, to cell 13 approximately doubled the rate of gas production increasing it to 7 mℓ/kg/day as compared to the value of 3.5 mℓ/kg/day observed before adding the simulated rainfall and buffer. Cell 14 received the same amount of rainfall as cell 13 but, in addition, it received recirculated leachate to simulate conditions in a solid-waste landfill which uses leachate recirculation. The leachate collected at the bottom of cell 14 was sprayed on top of a simulated landfill cover to allow a portion of the leachate to evaporate. The leachate that

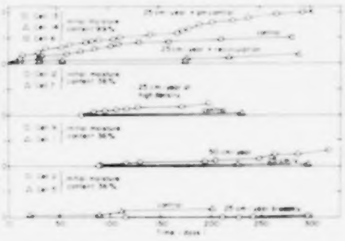

Figure 3. Cumulative Gas Production of the Enclosed Solid Waste During Subsequent Transient-State Testing

drained through the simulated cover was then mixed with the simulated rainfall and added to the top of cell 14. The addition of leachate and rainwater to the solid waste resulted in a decrease in gas production, i.e., it remained at a low level of approximately 1 mℓ/kg/day. At 200 days after the start of the transient period the gas production was gradually increasing. Control cell 6, which was always maintained at a 99% moisture content, continued to produce gas at a rate between that of cell 13 and 14 and showed a gradually decreasing rate.

The gas produced in the first 300 days by all three of these high-moisture containers is primarily carbon dioxide. However, cell 13 (the container receiving the buffer) started producing traces of methane in addition to carbon dioxide. However, after 360 days the composition of methane in gas produced from cell 13 increased to 62%. Although the pH of the leachate collected from cell 13 was similar to that from cell 14, i.e., 5.5, the pH of solid wastes at the upper portion of these drums might be quite different. The fact that high methane concentration was produced after a long period of 360 days of tansient-state addition of buffered simulated rainfall indicates a sufficiently large amount of buffer is required to initiate methane fermentation.

Lesser quantities of gas were produced in the four-cells in which the solid waste was initially maintained at 36% moisture (Figure 3). Addition of simulated rainfall at a rate of 25 cm/year was observed to generate leachate at the bottom of cell 12, containing high-density solid waste (485 kg/m^3 dry density), after a 145-day period, of cell 1 after 200 days, and of cell 2 after 247 days. Cell 9, which received rainfall at 50 cm/year, produced leachate after a 120-day period. The pH of leachates produced varied between 5.5 to 5.6 whereas the CODs were all slightly exceeding 100,000 mg/ℓ. The gas production in these cells generally started to increase before the solid waste had reached field capacity and before the leachate started to break through, indicating that a mositure content less than the field capacity is sufficient to produce increased gas production.

Before rainfall was added to cell 12, the container had ceased to produce any gas. Adding simulated rainfall caused gas production to resume after a 70-day "acclimation" period, indicating that the addition of water is capable of initiating gas generation. A similar observation can be made about the gas production in cells 1, 2 and 9. It is interesting to note that cell 9, which received the greatest amount of rainfall, did not produce twice as much gas as the containers receiving 25 cm/year, possibly indicating that the initial moisture content has a greater effect than the amount of moisture added at a later date. However, gas production from cell 2 and 12 (Figure 3) started increasing after 360 days of transient-state addition of 25 cm/year of simulated rainfall. The results of gas analysis showed that the oxygen concentrations in gases produced from cell 2 and 12 were respectively 11% and 13% with the balance being carbon dioxide. This indicated a possible leak of air into these cells although apparent leaks of these drums were not detectable. Inspite of this, additional research will be necessary to determine the relative importance of the initial moisture adjustment as compared to a gradual mositure increase.

The effects of environmental variables such as moisture content, temperature, size, density, and exposure to air, as evaluated in the present study, were compared with similar variables tested in other studies. The importance of moisture content in gas production, for example, is confirmed by the results of other studies (10, 11, 12). As shown in Figure 4, those results may indicate that a logarithmically increasing gas production can be realized by a linear increase in moisture content. The plotted rates of the present study represent those volumes produced during the initial 20 to 50-day monitoring period when the highest

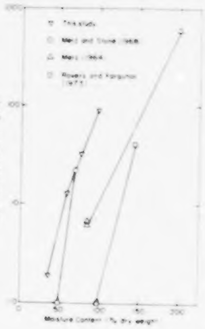

Figure 4. Effect of Moisture Content on the Rate of Solid Waste Gas Production as Noted in Different Studies

rates were observed. The values given here, therefore, tend to be higher than those from other studies.

In the present study, methane was detected only in the container with the large solid waste (cell 4) and in the container receiving buffer (cell 13). Using shredded solid waste fractions, Merz (12) observed a maximum CH_4 content of only 0.9 percent despite the fact that a wide range of moisture contents were tested. Even in the large solid waste container, Rovers and Farquhar (10) noted a methane content of only 2.8 percent. Ramaswamy (13), however, often noted stable methane fermentation as soon as 40 days after initiation of the tests. Careful examination of his data shows that methane fermentation does not start until the pH of the solid waste moisture increases beyond 5.0. The fact that methane fermentation was found only in that study may well be a result of the waste's high food content, which would have resulted in the release of sufficient amounts of ammonia during the degradation of the amino acids to counteract the pH decrease resulting from the release of free volatile fatty acids generated during the acid hydrolysis. The generation of methane in the container with the large solid waste may be a result of the relatively slow release of the volatile fatty acids, preventing the inhibition of the methane-generating bacteria.

Augenstein et al. (7) observed a high yield of methane gas, 128 ℓ/kg dry weight, using unstirred reactors for digestion to fuel gas of shredded municipal solid waste and wastewater sludge at a solid content of 48%. This is equivalent to 105% moisture content on dry solid basis. The high yield of methane gas production was, however, accomplished by employing finely divided calcium carbonate as a pH buffer. This collaborates with our results obtained with the addition of buffered simulated rainfall (cell 13). The fact that methane yield from cell 13 obtained from this study is only a small fraction of what observed by Augenstein et al. (7) indicates that pH and buffer conditions play a key role in the initiation of methane production. The concentration of free volatile fatty acids in leachate would also seem to play a role in methane production from the simulated landfills.

The results in Figure 5 clearly indicate that an increasing moisture content results in a larger percentage of methane in the gas phase and a higher rate of methane generation. However, in several studies discussed earlier, no methane was observed at the highest moisture content. Thus, the way in which the moisture interacts with the solid waste may determine whether methane generation will or will not commence. It would therefore seem beneficial to add buffering substances such as dewatered

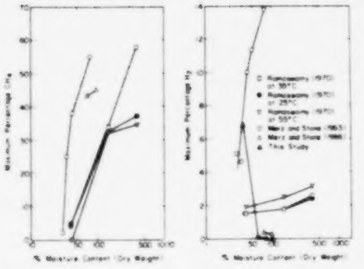

Figure 5. Effect of Moisture Content on Percentage of Methane and Hydrogen in the Gas Produced by Enclosed Solid Waste

anaerobic digester sludge, industrial alkaline sludge, spetic tank pumpings, or lime sludges to initiate methane generation.

Using the temperature data obtained in the present study and those from Ramaswamy (13), it is possible to calculate the activation energy of the reaction, defined as:

$$\ln k_2/k_1 = E/R\ (1/T_1 - 1/T_2)$$

where

k_1, k_2 = reaction constants at temperature T_1 and T_2

E = activation energy

R = gas content

The magnitude of the results shown in Figure 6 indicate that the rate of gas production, which parallels the hydrolysis of cellulose, is chemically controlled and not diffusion limited above a moisture content of about 75%, i.e., at an activation energy higher than 10 Kcal/g mole. Substantially increased rates of gas production are also at moisture contents greater than 75%. Moisture contents above 100% are not recommended because leachate will start to appear in the bottom of the landfill and may contaminate groundwater.

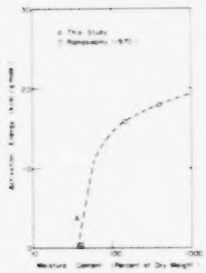

Figure 6. Effect of Moisture Content on Activation Energy of Solid Waste Placed at Different Temperatures

It should be realized that the generation of gas occurs only under anaerobic conditions; aerobic degradation of solid waste does not produce a large amount of gas because of the synthesis of biomass. It is also known that methane bacteria are obligate anaerobic, and exposure to oxygen will inhibit their action. This fact is clearly illustrated by data generated by Stone (14), who measured gas composition in leaky containers filled with solid waste and sludge mixtures. The data in Figure 7 indicate that the methane content of the gas phase decreases with respect to the carbon dioxide content when the percentage of nitrogen in the gas phase increases to 80%; i.e., when it becomes equal to the atmospheric content of this element.

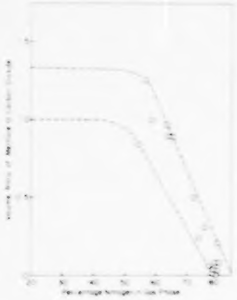

Figure 7. Effect of Nitrogen Content on the Ratio of Methane to Carbon Dioxide

CONCLUSIONS

The present study measured the rate and composition of gases released during anaerobic degradation of solid waste. The major gases observed were CO_2, H_2, and CH_4. Methane was produced in the container with the large waste particles and in the cell receiving buffering substances. The moisture content was the most important variable influencing the rate of gas production. It is recommended that landfills to be used for gas production be maintained at a moisture content above 75% but below 100%, or field capacity, to maximize gas production while holding the generation of leachate to an acceptable minimum, thereby preventing groundwater contamination. If landfills are to be used for methane production, co-disposal of refuse with buffering materials such as digested sludge, softening sludge, industrial alkaline sludge etc., and maintaining anaerobic conditions are recommended.

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LEACHATE AND GAS PRODUCTION UNDER CONTROLLED MOISTURE CONDITIONS

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ABSTRACT

In late 1974, a study was initiated to simulate various types of landfills. The intent of this program was to determine the effects on landfill behavior of varying moisture infiltration rates, adding pH buffering compounds, prewetting the wastes, varying ambient temperatures, and codisposing refuse with sewage sludge and various industrial wastes. A total of 19 landfill simulators were initially constructed. Each was loaded with approximately 3.2 metric tons of municipal refuse. Twelve were surcharged with small quantities of various industrial wastes and other materials. Maintenance, operation and monitoring of these landfill simulators was then begun. Water additions are made on all cells on a prescribed schedule. Gas and leachate are collected, quantified and analyzed. Refuse, soil and air temperatures and refuse settlement are recorded.

This paper addresses data derived from the municipal solid waste only test cells over approximately a 4-year period from November 1974 to August 1978. Test cell moisture balances are quantified and evaluated. Mass releases of various contaminants in leachate are plotted and scrutinized. Landfill gases are also addressed in terms of their components and total quantities.

INTRODUCTION

Under the Resource Conservation and Recovery Act of 1976 (RCRA), an inventory of all solid waste disposal sites is to be made across the nation. In addition, all sites identified by the inventory must be classified as either a "sanitary landfill" or an "open dump". Sites found to be in compliance with the Classification Criteria for Solid Waste Disposal Sites(1) (and thus posing no significant threat to human health or the environment) will be classified as sanitary landfills. Sites found to be in non-compliance with the criteria will be classified as open dumps. Open dumps must be either closed immediately or upgraded as necessary to mitigate the hazard identified by the inventory. The closure and upgrading procedures required will depend in large measure on the degree and status of the hazard. That is, if the hazard is large and yet to be fully exercised expensive confinement and/or waste removal procedures may be dictated. If the hazard is large but most of the damage is already done, money would be better directed toward clean-up than confinement and/or removal. Lastly, if the hazard is small, the site may be simply closed and funds used elsewhere.

A framework to assist in these decisions needs to be established. From numerous previous studies, data for such a framework has been generated on the decomposition processes in landfills and the release of contaminants from landfills into the environment. However, many of these efforts have had relatively short-lived monitoring terms. In addition, the variability and unknowns of initial conditions among such studies have made comparisons of data and compilation of large data bases difficult and often impossible.

This project was motivated by the need for such a data base. The broad objective of the program was to study solid waste decomposition and contaminant release at various types of landfills. Specific objectives are to determine:

1. The effect of different water infiltrations on solid waste decomposition.

2. The effect of sewage sludge additions on solid waste decomposition.
3. The effect of pH buffer addition on solid waste decomposition.
4. The effect of six selected industrial sludge additions on solid waste decomposition.
5. The effect of initial water addition on solid waste decomposition.
6. The survivability of poliovirus in landfills.
7. The effect of different air and soil ambient temperatures on solid waste decomposition.
8. The ability to duplicate monitoring data from two test cells constructed and operated under similar conditions.

TEST CELL CONSTRUCTION

The experimental test facility for this project is located at the U.S. Environmental Protection Agency's Center Hill Laboratory in Cincinnati, Ohio. A total of 19 test cells were constructed for this effort in November 1974 and April 1975. Fifteen of these are located outdoors and below the ground surface. These exterior test cells are arranged in a horseshoe alignment as shown in Figure 1. The remaining 4 cells are located inside the high bay area of the Center Hill Laboratory and are above ground resting on the concrete slab.

Individual test cells consist of steel tubes 1.83 m in diameter, 3.66 m high and 4.76 mm thick. Steel sidewalls were coated with coal tar epoxy as a moisture seal. The outside test cells were placed on concrete slabs in a excavated area. Soil was then backfilled around the sidewalls to within 0.3 m of the top of the cell. Several layers of fiberglass cloth were placed inside at the base of these cells and extended 0.3 m up the sidewall to provide a watertight and gas tight seal. The interior test cells were placed atop steel bases welded into place.

Provisions for leachate drainage were installed for all cells. Exterior cells had a small depression in the concrete slab and connective piping to a leachate collection well. This well serves as a central leachate collection point for all exterior cells. It is also used as a groundwater drawdown to prevent pressure and infiltration of groundwater into the test cells. Interior cells are mounted on concrete blocks and leachate drains are attached below the cell bottoms.

The test cells were then readied for waste loading. To minimize the exposure time and more closely simulate actual landfill conditions all test cells were completely loaded and covered within 7 days in each loading period. First, a 15.2 cm thick layer of silica gravel was applied at the bottom of each cell. This serves as a reservoir for leachate and prevents refuse from clogging the drain. Silica gravel was

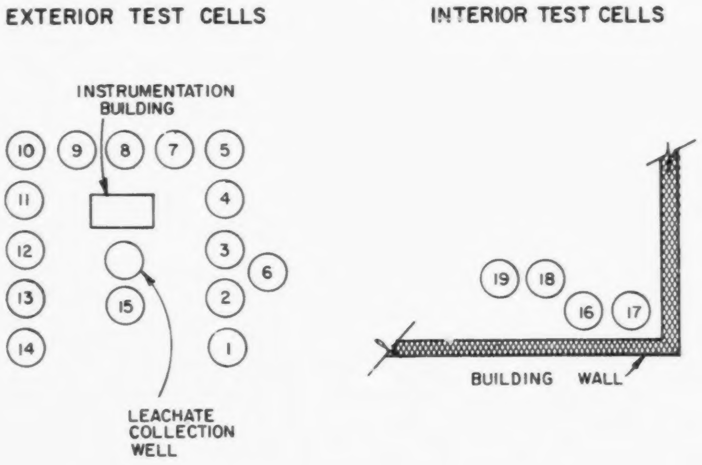

Figure 1. Test Cell Location Plan

selected to prevent any chemical reaction with collected leachate. Refuse was then delivered to the site and added in eight 0.3-m thick increments or lifts. Each lift was compacted with a wrecking ball to a density of approximately 270 kg/m^3. Sludges and other materials added to selected cells were applied to the top of each lift (except the first) in proportionate amounts. No additives were applied to the first lift to avoid any premature leaching of moisture. Temperature probes were installed atop the second, fourth and sixth lifts in each cell. Gas probes were installed atop the second and sixth lifts.

At completion of waste loading a 0.3-m thick layer of silty clay cover soil was applied atop the waste. An additional 0.3-m thick layer of pea gravel was placed atop the cover soil, and a gas probe and water distribution ring installed in the gravel. A settlement indicator was affixed to the top of the gravel and mounted inside a sight glass through the steel lid on each cell. This lid is bolted down to the test cell and caulked to provide an airtight seal. A cross-section of a typical test cell is shown in Figure 2.

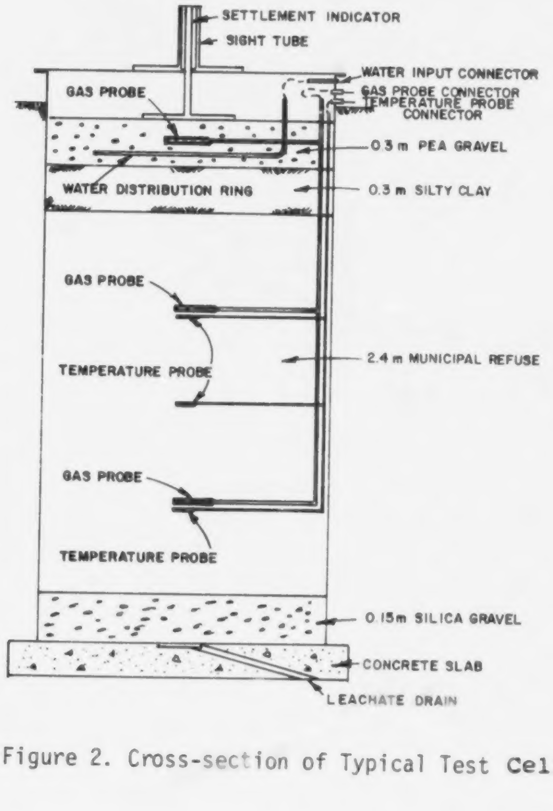

Figure 2. Cross-section of Typical Test Cell

WASTE CHARACTERIZATION

As mentioned previously, sludges and other materials were added to selected cells. These additives are identified in Table 1 along with other pertinent initial loading conditions such as weight and moisture contents. The intent of these additions was to allow an investigation of what effects codisposing these materials with municipal refuse would have on solid waste decomposition. More specifically, this arrangement provided test cells which fulfilled the earlier objectives as follows:

1. Different water infiltration rates: Test Cells 1, 2, 3 and 4
2. Sewage sludge additions: Test Cells 5, 6 and 7
3. pH buffer addition: Test Cell 8
4. Six selected industrial sludge additions: Test Cells 9, 19, 12, 13, 14 and 17
5. Initial water addition: Test Cell 11
6. Survivability of poliovirus: Test Cell 15
7. Different ambient air and soil temperatures: Test Cells 2 and 16
8. Duplication of monitoring data: Test Cells 16, 18 and 19

Before loading, all solid waste for each lift in each cell was weighed and sorted. Eleven sort categories were used and the average composition in each cell was then computed. A summary of the solid waste composition of each cell is shown in Table 2. In addition to categorization, two samples were extracted from each lift. An 11.4 kg sample was used for moisture content determination; these were then averaged over each cell to determine the moisture contents shown in Table 1. An 18.2 kg sample was used for chemical analysis; results were computed for each sort category. A sample composited according to average sort categories in all cells was also chemically analyzed (see Table 3).

OPERATION AND MONITORING

To more closely simulate actual infield environmental conditions, all test cells are operated in accordance with a strict monthly schedule. First, gas systems on each cell are controlled by top positions. As shown in Table 1, tops may be (1) permanently open as for Test Cells 9, 10, 12, 13

TABLE 1. TEST CELL LOADING AND OPERATION

Cell Number	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19
Refuse																			
wet weight (kg)	3025	2989	3007	3002	3001	2919	2964	2994	3001	2998	2924	3048	3006	3015	1010	2996	2998	3000	3012
moisture content (% of ww)	35	35	35	35	35	35	35	35	32	32	35	35	35	35	32	35	34	32	32
initial moisture (l)	1056	1043	1049	1048	1047	1019	1034	1045	966	965	1020	1064	1049	1052	969	1046	1046	966	970
dry weight (kg)	1969	1946	1958	1954	1955	1900	1930	1949	2035	2033	1094	1984	1957	1963	2041	1950	1972	2034	2042
Additive																			
type	--	--	--	--	sewage sludge	sewage sludge	sewage sludge	$CaCO_3$	petroleum sludge	battery waste	H_2O	electro-plating waste	inorganic pigment waste	chlorine prod. brine sludge	polio-virus	--	solvent based paint sludge	--	--
wet weight (kg)	--	--	--	--	68	204	680	90.7	1518	1291	1243.2	1190.4	1420.6	2038.7	--	--	1604.0	--	--
moisture content (% of ww)	--	--	--	--	88	88	88	10	79	89	100	80	51	24	--	--	25	--	--
initial moisture (l)	--	--	--	--	60	179	598	9	1199	1153	1293	946	728	491	--	--	1206	--	--
dry weight (kg)	--	--	--	--	8	25	82	82	319	138	0	244	693	1548	--	--	1206	--	--
Solid Waste Total																			
wet weight (kg)	3025	2989	3007	3002	3069	3123	3644	3084.7	4519	4289	4217.2	4238.4	4426.6	5053.7	3010	2996	4602	3000	3012
moisture content (% of ww)	35	35	35	35	36	38	45	34	48	49	55	47	40	31	32	35	31	32	32
initial moisture (l)	1056	1043	1049	1048	1107	1198	1632	1054	2165	2118	2313	2010	1777	1543	966	1046	1444	966	970
dry weight (kg)	1969	1946	1958	1954	1963	1925	2012	2031	2354	2171	1904	2228	2650	3511	2041	1950	3158	2034	2042
Top Type																			
open									x	x		x	x	x					
open/closed	x	x			x	x	x	x			x				x				
closed			x	x												x	x	x	x
Annual Infiltration																			
mm	203.2	406.4	609.6	812.8	406.4	406.4	406.4	406.4	406.4	406.4	406.4	406.4	406.4	406.4	406.4	406.4	406.4	406.4	406.4
l	533.30	1067.01	1602.95	2135.87	1067.01	1067.01	1067.01	1067.01	1067.01	1067.01	1067.01	1067.01	1067.01	1067.01	1067.01	1067.01	1067.01	1067.01	1067.01

TABLE 2. REFUSE COMPOSITION IN TEST CELLS (2)

Test cell	Food waste	Garden waste	Paper	Plastic, rubber leather	Textiles	Wood	Metal	Glass	Ash, rock and dirt	Diapers	Fines
1	9.5	13.8	37.1	6.2	5.0	1.4	12.2	9.3	3.3	2.5	4.0
2	6.3	21.5	41.5	5.4	4.5	2.2	8.5	9.9	3.3	1.2	3.5
3	11.0	30.2	34.9	8.3	3.3	.3	8.6	5.9	3.4	2.3	2.4
4	5.9	22.6	37.8	6.4	4.1	2.0	5.9	8.8	1.4	3.2	3.5
5	11.3	16.6	41.2	6.9	8.0	1.1	8.0	7.0	1.6	1.8	2.9
6	6.1	25.9	36.5	5.4	1.3	1.4	8.1	9.4	3.5	1.5	3.0
7	5.3	20.3	43.6	6.4	3.4	1.7	9.0	6.4	2.9	2.4	2.8
8	6.6	11.1	53.1	8.3	3.9	1.1	5.8	6.4	1.6	1.6	1.7
9	7.6	17.0	41.3	11.9	2.1	1.9	8.9	5.3	2.8	.9	2.1
10	11.2	8.2	44.0	6.1	4.0	4.4	8.8	7.2	4.7	2.9	3.3
11	8.7	16.2	39.8	5.5	5.6	1.6	8.4	9.0	2.0	2.4	3.4
12	8.2	9.5	46.5	5.3	2.7	1.2	7.7	8.8	6.1	1.5	4.2
13	6.3	20.7	39.1	5.7	4.4	1.4	8.3	7.8	1.8	3.7	2.9
14	9.0	11.1	37.1	6.8	5.6	1.5	9.9	8.2	4.1	4.8	4.1
15	12.7	8.4	42.4	7.3	3.5	2.9	7.1	7.9	4.3	2.8	3.6
16	7.3	16.3	41.8	10.1	3.8	.8	8.9	6.0	1.3	2.7	2.8
17	6.0	15.6	45.7	5.3	6.0	1.4	7.7	6.6	2.7	1.6	2.8
18	6.2	9.8	48.3	6.1	3.3	3.9	9.3	8.4	3.0	2.9	3.4
19	8.5	11.9	43.2	7.5	2.7	1.9	7.4	10.0	1.9	3.0	3.6
Avg.	8.1	15.7	41.8	6.9	4.1	1.8	8.3	7.8	2.9	2.4	3.2
Std. dev.	2.2	5.3	4.5	1.8	1.5	1.0	1.4	1.4	1.3	1.0	.66

* Percent by wet weight

TABLE 3. REFUSE CHEMICAL ANALYSIS (3)

Component	Paper	Garden	Metal	Glass	Food	Plastics, rubber, leather	Fines†	Ash, rocks, dirt	Diapers	Wood	Composite
COD§	0.804	0.815	0.492	0.011	0.754	2.14	0.935	0.040	0.720	0.503	0.520
TKN	0.028	0.171	0.022	0.140	3.09	1.25	0.131	0.119	0.138	0.228	0.247
Total Phosphate	0.048	3.14	2.79	0.049	10.4	1.40	1.97	4.48	2.65	0.103	2.32
Lipids	2.47	3.04	0.420	1.54	13.8	5.02	4.85	1.52	2.26	1.00	2.84
Ash	92.0	36.5	4.85	2.25	41.6	182	49.5	19.6	96.0	77.9	25.8
Crude fiher	21.7	16.6	0.235	0.040	10.5	21.5	6.39	5.85	13.7	20.8	11.3
Total carbon	58.0	14.4	4.80	0.750	19.5	15.8	16.4	13.4	44.5	51.0	24.8
Inorganic carbon	4.30	4.66	3.40	0.220	2.58	5.75	4.30	7.80	0.740	0.380	3.08
Organic carbon	53.7	9.74	1.40	0.530	17.0	10.1	12.1	5.60	43.8	50.7	21.8
Sugar as Sucrose	<0.1	1.71	<0.1	<0.1	6.08	<0.1	1.18	<0.1	<0.1	<0.1	3.50
Starch	3.40	7.42	<0.1	<0.1	8.57	3.42	7.20	6.40	<0.1	0.78	16.2
Asbestos	NA¶	NA	NA	NA	NA	NA	NA	NA	NA	NA	<1**
Arsenic††	<0.1	NA	<0.1	10.2	NA	NA	1.2	3.6	<0.1	<0.1	80
Selenium††	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	<0.1
Mercury††	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	5.0
Lead††	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	15.0
Beryllium††	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	<0.1
Cadmium††	0.36	NA	20.9	2.7	NA	1.8§§	4.2	4.5	0.25	1.6	2.4
Iron††	375	330	625*	3220	505	444	0.392*	0.340	99.0	0.378	0.782*
Zinc††	50.0	106	175	9.75	59.0	118	322	181	343	59.4	127
Chromium††	8.2	1.1	15.3	1.1	1.3	2.0	13.1	10.1	0.5	1.1	10.7
Manganese††	13.1	194	870	15.7	12.2	12.1	115	177	5.90	50.0	90.5
Potassium††	11.2	0.135*	1.00	2.70	0.162*	98.7	135	555	750	90.0	475
Magnesium††	160	4175	80.5	472	377	289	1.02*	2.63*	279	253	0.225*
Calcium††	77.5	0.830*	<0.25	16.2	0.465*	912	2.11*	4.08*	360	590	0.437*
Sodium††	9.70	185	37.0	60.0	804	143	400	0.209*	0.110*	572	950
Copper††	4.5	9.34	0.221*	2.54	8.58	12.4	35.8	32.6	4.14	38.2	31.6
Nickel††	15.7	15.7	115	19.0	12.5	32.0	33.2	10.1	3.36	27.0	10.1
Moisture	56.7	156.4	8.80	2.00	216.5	57.04	123	30.79	133	21.43	
Moisture¶¶	35.20	56.91	6.18	1.65	70.07	49.27	49.36	18.52	66.28	17.10	
Composition	42.6	10.7	12.2	12.2	3.6	8.7	2.9	3.2	1.3	2.6	
Composition¶¶	41.62	15.77	8.21	7.83	7.56	10.91	3.58	3.36	2.47	1.99	

* Percent by dry weight unless otherwise specified.

† <25.4 mm (1.0 inch).

§ g COD/g sample.

¶ Not analyzed.

** Fibers per gram.

†† Parts per million by weight.

§§ Plastics, rubber, and leather not analyzed.

¶¶ Percent wet weight.

and 14; (2) permanently sealed closed as for Test Cells 3, 4, 16, 17, 18 and 19; or (3) opened and closed in accordance with a monthly schedule as for Test Cells 1, 2, 5, 6, 7, 8, 11 and 15. The schedule of open/closed gas systems is shown in Table 4. These gas systems are closed only during selected months to simulate temporary in-field conditions of frozen or saturated ground cover.

TABLE 4. MONTHLY CELL OPERATION SCHEDULE

	Gas System Schedule	Infiltration Schedule				
			Amount per month (liters)			
Month	Position on Open/ Closed Cell Tops	Type per Month	203.2 mm Annual Rate	406.4 Annual Rate	609.6 Annual Rate	812.8 Annual Rate
January	Open	None	0	0	0	0
February	Open	Low	48.52	96.90	145.72	194.17
March	Closed	High	96.90	194.17	291.45	388.34
April	Closed	High	96.90	194.17	291.45	388.34
May	Open	High	96.90	194.17	291.45	388.34
June	Closed	Low	48.52	96.90	145.72	194.17
July	Closed	None	0	0	0	0
August	Open	None	0	0	0	0
September	Closed	None	0	0	0	0
October	Open	Low	48.52	96.90	145.72	194.17
November	Closed	Low	48.52	96.90	145.72	194.17
December	Closed	Low	48.52	96.90	145.72	194.17
Total	--	--	533.30	1067.01	1602.95	2135.87

Secondly, water is applied to each test cell in accordance with the annual infiltration rates listed in Table 1. As shown, with the exception of Test Cells 1, 3 and 4, all test cells receive an annual water application of approximately 400 mm. Test Cells 1, 3 and 4 receive annual applications of approximately 200, 600 and 800 mm, respectively. In this way, the goal of varying the moisture regimen is realized in Test Cells 1, 2, 3 and 4 (with 200, 400, 600 and 800 mm of infiltration, respectively). Water is applied in accordance with the schedule shown in Table 4. As shown, an attempt has been made to duplicate in-field conditions. Specifically, high applications are made in the normally rainy months of March, April and May. No applications are made in the normally dry months of January, July, August and September. Low applications are made in the remaining months.

Physical data recording, sampling and analysis are performed in accordance with the schedule in Table 5. To summarize this schedule, gas volumes, refuse temperatures and refuse settlements are recorded regularly. Gas samples are collected each month and analyzed via a gas chromatograph for five major constituents. Leachate samples are collected each month, their volumes recorded, and representative samples prepared. Monthly samples are then analyzed for 22 parameters. An additional 12 parameters are determined quarterly and 7 parameters semi-annually. All analyses are performed at the Center Hill Laboratory using facilities and equipment used exclusively for this project. Strict quality assurance program procedures are utilized to ensure the validity of sampling results. Analytical data derived from the project is punched onto computer cards and entered into the memory banks of an EPA in-house computer. This computational facility can then be used for plotting project data as an aid in evaluating results.

MOISTURE BALANCE EVALUATION

Previous papers on this project have addressed the behavior of the test cells which have been surcharged with industrial sludges and other additive materials. It is

TABLE 5. DATA RECORDING, SAMPLING AND ANALYTICAL SCHEDULE

SAMPLE	ANALYSIS	TEST CELL	FREQUENCY
Gas	CH_4	All	Monthly
	CO_2	All	Monthly
	N_2	All	Monthly
	O_2	All	Monthly
	H_2	All	Monthly
	Volume	16, 17, 18, 19	Daily
Soil/ Refuse	Temperature	1 - 15	Biweekly
	Temperature	16, 17, 18, 19	Daily
	Settlement	All	Monthly
Leachate	TOC	All	Monthly
	COD	All	Monthly
	Total Solids	All	Monthly
	Total Volatile Solids	All	Monthly
	pH	All	Monthly
	ORP	All	Monthly
	Specific Conductance	All	Monthly
	Alkalinity	All	Monthly
	TKN	All	Monthly
	ortho-PO_4	All	Monthly
	Fe	All	Monthly
	Cd	All	Monthly
	S^-	All	Monthly
	Cl	All	Monthly
	Hg	All	Monthly
	V	2, 4, 10, 13	Monthly
	Be	2, 4, 9, 10, 12, 13, 14	Monthly
	Se	2, 4, 9, 12, 13	Monthly
	CN	2, 4, 10, 13	Monthly
	Phenol	2, 4, 5, 6, 7, 9	Monthly
	As	2, 4, 14	Monthly
	Cr^6	2, 4, 9, 13	Monthly
	Cu	All	Quarterly
	Pb	All	Quarterly
	Ni	All	Quarterly
	Zn	All	Quarterly
	Asbestos	2, 4, 5, 6, 7, 13, 14	Quarterly
	Sn	All	Quarterly
	Ti	2, 4, 14	Quarterly
	Total PO_4	All	Quarterly
	Organic Acids C_2 to C_6	All	Quarterly
	Fecal Coliform	All	Quarterly
	Fecal Streptococci	All	Quarterly
	Cr total	All	Quarterly
	As	All	Semi-annually
	Phenol	All	Semi-annually
	Be	All	Semi-annually
	Ti	All	Semi-annually
	B	All	Semi-annually
	V	All	Semi-annually
	Sb	All	Semi-annually

the intent of this paper to concentrate on the behavior of the municipal solid waste only test cells.

First, the moisture balance of Test Cells 1, 2 and 4 will be considered. As stated previously, these test cells were established to evaluate the effect of a varied rainfall regimen. They receive 200, 400 and 800 mm of annual infiltration, respectively. Figures 3, 4 and 5 have been included to demonstrate the cumulative quantities of moisture added and leached from each cell. These figures could also be useful in serving as design charts. For a given annual infiltration of 200, 400 or 800 mm, the total moisture added (in l/kg) at any point in time can be identified from the upper line on each chart. The total moisture leached (also in l/kg) can be identified from the lower line on each chart.

Several things need to be kept in mind when using these charts. First, these charts are based on a 2.4 m deep landfill. For the same infiltration rate, greater depths of fill would require an appropriate downward adjustment to both the moisture added and the moisture leached. Second, the annual rates of 200, 400 and 800 mm are for infiltration, not precipitation. Precipitation is greater than infiltration; runoff and evapotranspiration must be subtracted from precipitation to yield infiltration.

Some observations can be made from Figures 3, 4 and 5. First, it is obvious that the cumulative amount of moisture leached is considerably less than the moisture added for any point in time. This is due to the ability of refuse to retain a certain quantity of moisture. Generally, if the initial moisture of the refuse plus the

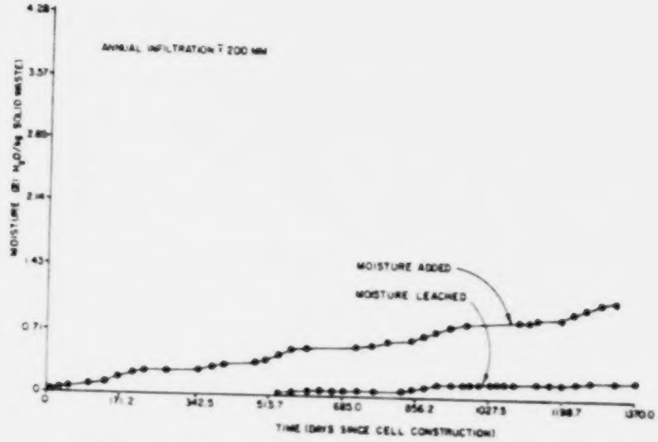

Figure 3. Moisture At Test Cell 1

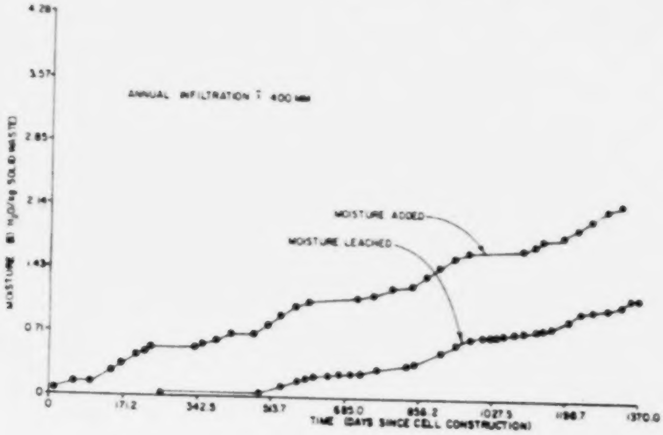

Figure 4. Moisture At Test Cell 2

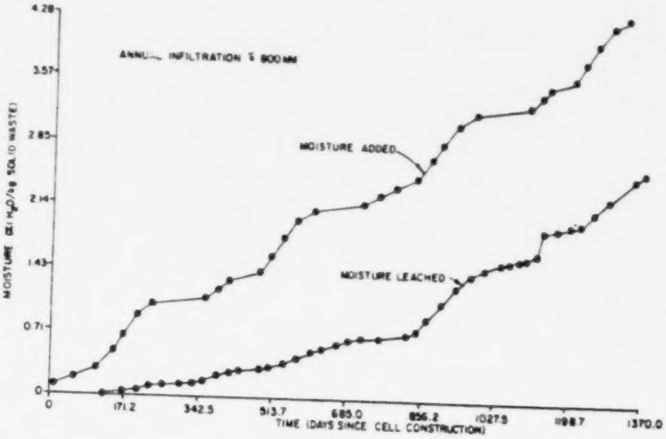

Figure 5. Moisture At Test Cell 4

moisture added does not exceed this retention ability, leachate will not appear. This quantity is known as the "field capacity" and is technically defined as "the maximum moisture content which a soil (or solid waste) can retain in a gravitational field without producing continuous downward percolation."(5) Generally too, the removal of significant quantities of leachate marks the initiation of continuous leaching and the point at which this field capacity is attained. Some leachate may be emitted before this point; but generally, these are small quantities and may be followed by times at which no leachate is emitted. As shown in Figures 3, 4 and 5, the field capacities for infiltration rates of 200, 400 and 800 mm were attained at 540, 480 and 150 days, respectively. Moisture conditions present in each test cell at this initiation of continuous leaching are detailed in Table 6. The equation to define the water balance in each cell at that time is:

$$\text{Moisture Retained} = (\text{Initial Moisture}) + (\text{Moisture Added}) - (\text{Moisture Leached})$$

The moisture retained in each cell at the point of initial continuous leaching was found to exist within a fairly tight range of 1.02 to 1.31 l/kg on a dry weight basis. (All mass values from this point on are based on dry weights.) Presumably, after field capacity has been reached, one liter of moisture should be leached for each one liter of moisture added. Thus, the moisture retained should stabilize after the onset of continuous leaching. However, this has not been the case. As shown in Figures 3, 4 and 5, the gap between cumulative moisture added and cumulative moisture leached has stabilized somewhat with the onset of initial continuous leaching; however, the size of the gap is growing even larger. This gap is quantified in Table 6. As shown, Cells 1, 2 and 4 now retain between 1.45 and 2.23 l/kg. The reason for this increase cannot be fully explained. One possible reason, however, may be the effect of channeling. Channeling exists when moisture added to a waste percolates directly through it, failing to fill all the voids before leaching. Some voids remaining in the waste may gradually be retaining moisture rather than letting it channel through; this is one possible explanation for an increasing moisture retention beyond the point of continuous leaching. (Other possible explanations could be failure to collect all leachate generated due to an inordinate time for collection or perhaps even a leak in the collection system.)

LEACHATE QUALITY EVALUATION

Approximately 41 parameters are determined on leachate samples collected for this project. Eleven of these parameters were selected for in-depth investigation. It is appropriate to note that graphs included herein are plots of mass leached (cumulative kg of contaminant/kg of dry solid waste) as a function of leachate volume (cumulative l of leachate/kg of dry solid waste). An attempt was first made to plot mass leached vs. time. It was thought that this would be more meaningful to the designer. However, generally histograms are somewhat irregular (due to seasonal fluctuations in applied moisture) and thus trends are not as readily discernible. To make the plots in this section useful as a design aid, it is suggested that the user consult the previous plots (Figures 3, 4 and 5) to determine expected leachate volume for a given infiltration rate and time. Once expected leachate volume is computed, contaminant release can be determined from the ensuing plots in this section.

TABLE 6. MOISTURE BALANCE SUMMARY

TEST CELL	ANNUAL INFILTRATION		MOISTURE TO CONTINUOUS LEACHING					MOISTURE TO DATE				
	(mm)	(l/kg)	No. of Days	Initial Moisture (l/kg)	Moisture Added (l/kg)	Moisture Leached (l/kg)	Moisture Retained (l/kg)	No. of Days	Initial Moisture (l/kg)	Moisture Added (l/kg)	Moisture Leached (l/kg)	Moisture Retained (l/kg)
1	203.2	0.27	540	0.54	0.49	0.01	1.02	1370	0.54	1.06	0.15	1.45
2	406.4	0.55	480	0.54	0.80	0.03	1.31	1370	0.54	2.14	1.11	1.57
4	812.8	1.09	150	0.54	0.70	0.05	1.19	1370	0.54	4.27	2.58	2.23

pH

A plot of pH as a function of cumulative leachate volume is included as Figure 6. As shown, all three curves exhibit somewhat similar trends. Generally, pH values cycled over a wide range between 4.7 and 7.6 in the early leaching stages (between 0 and 0.32 l/kg of cumulative leachate volume). The average pH during this phase is seen to be about 5.6. After this period, pH values continue to exhibit a cyclic nature, but over a more confined pH range between 4.8 and 5.6. The average pH during this phase is seen to be about 5.2. The large fluctuations and higher average value in the early phase are to be expected. Early on in leachate generation, anaerobic conditions are incomplete. Where they do exist, pockets of organic acids may be generated and these may come through as slugs in the leachate, greatly reducing the pH. Most of the infiltrating water during this period, however, merely channels through the waste without picking up the acids. The result is a leachate pH similar to that of the infiltrating water. Later on in leachate development, field capacity is approached or even attained, reducing the occurance of channeling. The effect is more thorough mixing, creating a well-buffered leachate. In addition, more complete anaerobic conditions prevail. As a result, organic acids are generated in greater quantities, driving the average pH to lower values.

TOC

The release of TOC appears to be approaching an asymptote (Figure 7) at which only nominal amounts of TOC will subsequently be released. This trend is most apparent for Test Cell 4 (at 800 mm/year infiltration) which has yielded the greatest cumulative leachate volume. The curve for Test Cell 2 exhibits a slightly higher trend and thus, greater TOC release for a given leachate volume. This tendency for greater TOC release with lower infiltration rates is not supported by data from Cell 1; however, the data base from Cell 1 is too limited to support conclusions about its ultimate trending.

Despite these observations, there is generally no substantial difference apparent among the curves for these three cells. That is, cumulative TOC leached for a given cumulative leachate volume appears to be the same regardless of the infiltration rate. Thus, the effect of a higher infiltration rate is merely to accelerate the release process with time. Ultimately, it would appear, each cell will have similar overall TOC release.

As shown in Table 3, the TOC in the solid waste, when loaded into the test cells was determined to be 21.8% by dry weight. Since 31,800 mg/kg of TOC have been leached by Cell 4 to date, 14.6% of the initial TOC has been leached by this Cell. This percentage leached is illustrated on the right vertical axis in Figure 7. Since the overall trend of the TOC curve appears to be asymptotic, it would be presumed that ultimate TOC leaching would not be much larger than this value. The balance of the initial TOC may remain in the solid waste and never be made available for

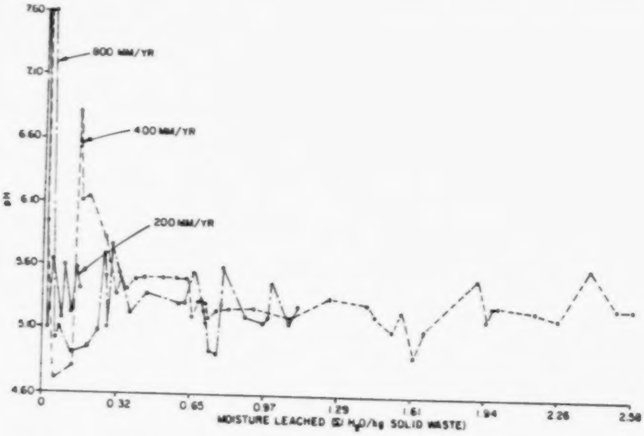

Figure 6. Moisture Leached: pH

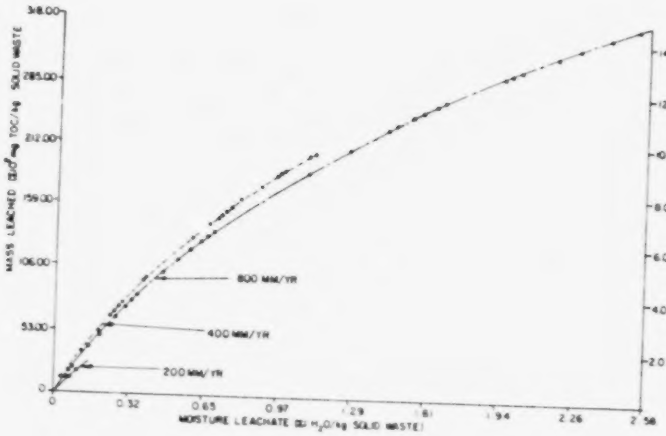

Figure 7. Mass Leached: TOC

leaching. Alternatively, some of the organic carbon may be converted by microorganisms into gases (CO_2 and CH_4). A portion of the CO_2 may then be expected to be leached as a carbonate.

COD

COD removal (Figure 8) for the 400 mm/year cell exhibits the highest trajectory, although a recent trend has caused it to dip below the 800 mm/year cell. The trend for the 200 mm/year cell is again lower than the two higher infiltration cells. All three cells follow a similar COD pattern.

From observations, the asymptotic-trending curve for COD appears to have the same slope as that for TOC. Nevertheless, it is likely that the COD curve will climb at a faster rate (i.e., decline at a slower rate) in the future. Initially, TOC is one of the primary contributors to the COD determination. However, inorganics also contribute to COD and as TOC tapers off these may become the driving force behind COD. The cyclical behavior of pH is especially conducive to the continued appearance of COD in leachate. When pH values dip low and CO_2 is available in the absence of O_2, ferrous iron and other inorganic contributors to COD are created and held in solution.

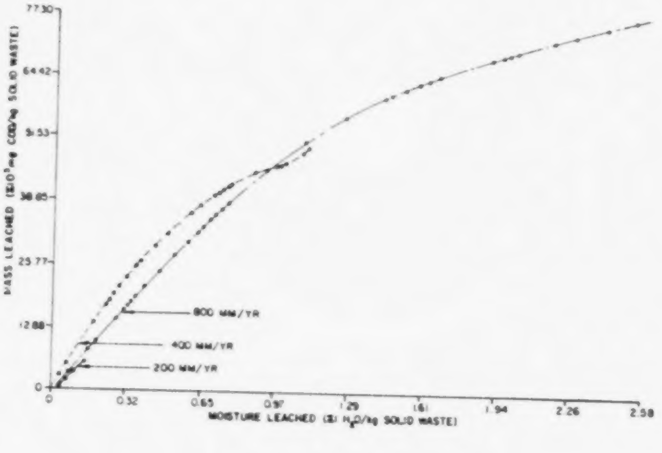

Figure 8. Mass Leached: COD

Total Solids

Total solids release (Figure 9) also reveals an asymptotic trending. The curve for Cell 2 reveals a greater divergence from that for Cell 4 than was encountered previously. Nevertheless, the overall trend for Cell 2 is similar to Cell 4. As shown in Table 1, each of these cells had a total solids dry weight of slightly less than 2000 kg. Thus, for Test Cell 4, when 42,200 mg of solids have been leached to date, approximately 4.2% of available solids has been released. The asymptotic trend of the curve would indicate that ultimate release of solids with time would not be much greater than this amount.

The curve for total solids reveals a slightly greater slope than that for TOC or COD. This is to be expected since release of solids (ultimately in the form of inerts) is expected to continue long after release of TOC and COD have tapered off to nominal quantities. The nature of the solids released could be expected to vary considerably with time. Initially, solids released are probably inorganic dusts and readily soluble inorganics present in the solid waste which are being flushed out in the leachate. The solid waste sort categories contributing to these dusts are likely to be the fines, ground glass, ash, rock and dirt. After this effect is completed, organic contributions increase and may reach a 50 - 50 level with inorganics in the solids. Ultimately, inorganics prevail in the total solids once again.

TKN

Total kjeldahl nitrogen leached by Cells 1, 2 and 4 is plotted in Figure 10. Again, the release exhibits an asymptotic trending for all three infiltration rates. This time, the release plotted for both the 400 mm and the 200 mm cell exhibit a higher trajectory than the 800 mm cell. The deviation is greater among the three curves than was found for COD and TOC and is similar to that found for total solids. As shown on the right axis, 74.1% of the TKN assayed in the solid waste has been leached. This value is much higher than has been seen in the release of other parameters. However, high TKN release is expected since conditions are ideal for biological denitrification as a result of high temperature, acid pH and the presence of organics. Whether over 70% has actually been released is questionable, however. It is possible that the initial TKN reading in the solid waste is erroneous due to faulty analysis or compilation of a solid waste sample that was not representative of TKN actually in the waste.

Metals: Cd, Fe, Hg, Ni, Pb and Zn

Mass release for six selected metals are presented in Figures 11 through 16. Generally, these curves can be classified into three types. The first type is the familiar asymptotic trend seen for most of the previously described parameters. This trend is exhibited by mercury, nickel and zinc in Figures 13, 14 and 16, respectively.

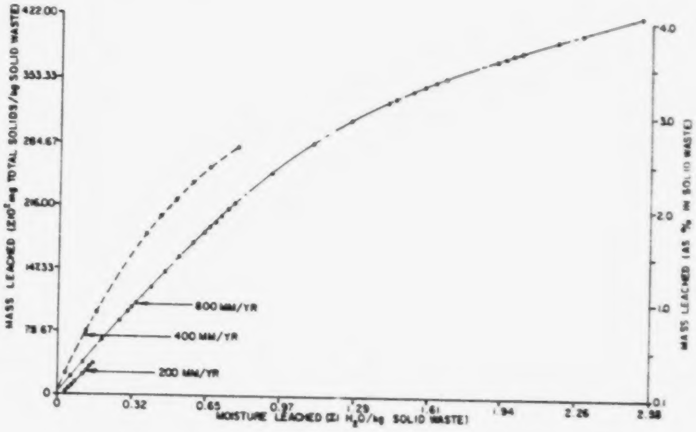

Figure 9. Mass Leached: Total Solids

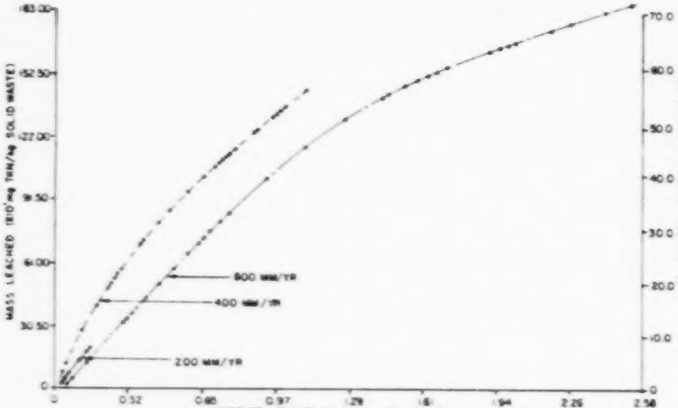

Figure 10. Mass Leached: TKN

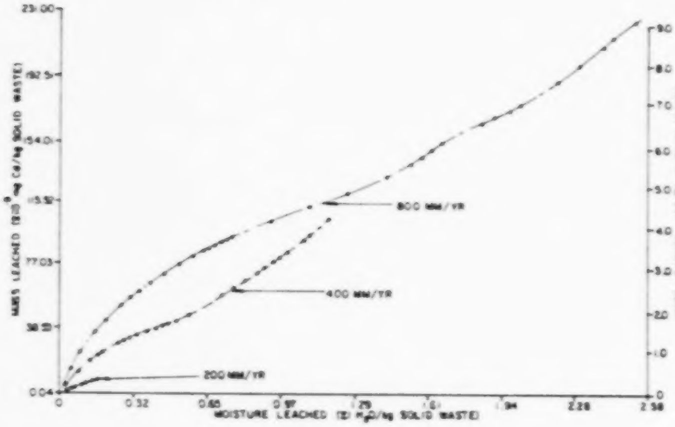

Figure 11. Mass Leached: Cd

As was generally the case for previous asymptotic curves, the trend for the 400 mm/year cell has a higher release than that for the 800 mm/year cell. The 200 mm/year fluctuated among being below, between and above the release trend for the other two cells for mercury, nickel and zinc, respectively.

Mass release to date as a percentage of that available for nickel was found to be 15.6%, similar to TOC. Mercury leached, on the other hand, amounted to only 0.9% of that available; this was less than any other parameter investigated. Zinc, however, was found to have leached 90.6% of that assayed in the refuse; this was far greater than for any other parameter investigated. Release of zinc would be expected to be high since it has very high solubility relative to the other metals investigated here. (Nonetheless, it is unlikely that over 90% of zinc has been leached. Possible explanations may again be inaccurate analysis or a non-representative solid waste sample.) The behavior of nickel and zinc as asymptotic curves is somewhat surprising. Most metals examined here are governed by chemical reactions and could be expected to yield a constant release and straight-line plot. Mercury is the one possible exception to this since biological reactions may predominate in the creation of methyl mercury.

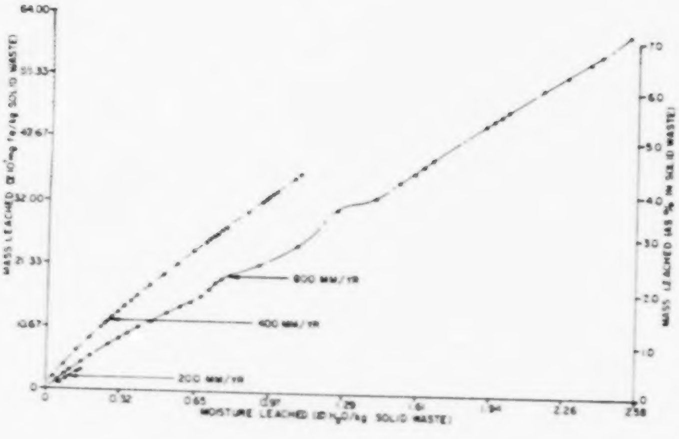

Figure 12. Mass Leached: Fe

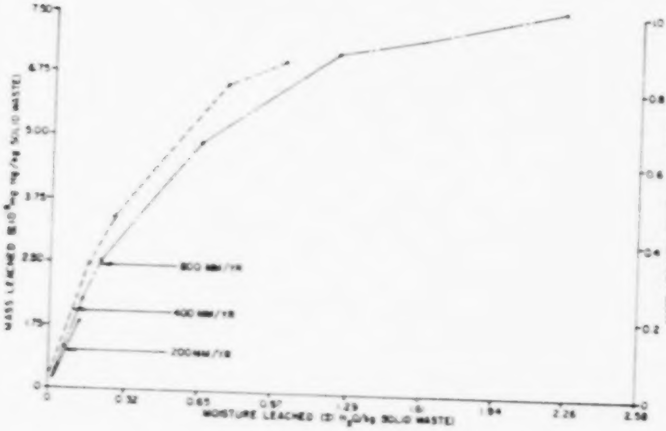

Figure 13. Mass Leached: Hg

The second type of curve, a linear relationship, was observed for cadmium and iron in Figures 11 and 12, respectively. In each case, the trajectories for the 400 mm and 200 mm/year cells exhibit the same type of straight line behavior as for the 800 mm/year cell. However, the 400 mm/year cell plot in the cadmium figure exhibits a lower trajectory than that for the 800 mm/year cell. The 400 mm cell plot for iron exhibits its usual higher trajectory than the 800 mm cell. The percent of available mass released is similar for both these metals at 9.6% for cadmium and 7.7% for iron. However, owing to the apparent nondeclining (straight line) growth trends observed for cadmium and iron (relative to the declining asymptotic trends seen elsewhere) ultimate release of these two metals could be expected to greatly exceed these percentages.

The third type of curve observed among these plots is exhibited by lead in Figure 15. This plot shows an initial linear release, followed by a sharp increase. This trajectory change occurs at a total leachate volume of approximately 1.61 l/kg for the 800 mm/year cell. Oddly, however, this change occurs at a leachate volume of about 0.65 l/kg for the 400 mm/year cell. The reason for this behavior is unexplained; generally before, we have seen similar trajectory behavior and changes in trajectory to occur at a single total leachate volume, regardless of the infiltration rate.

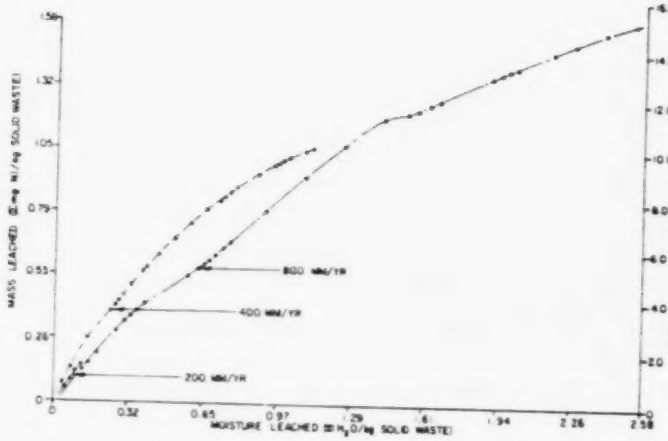

Figure 14. Mass Leached: Ni

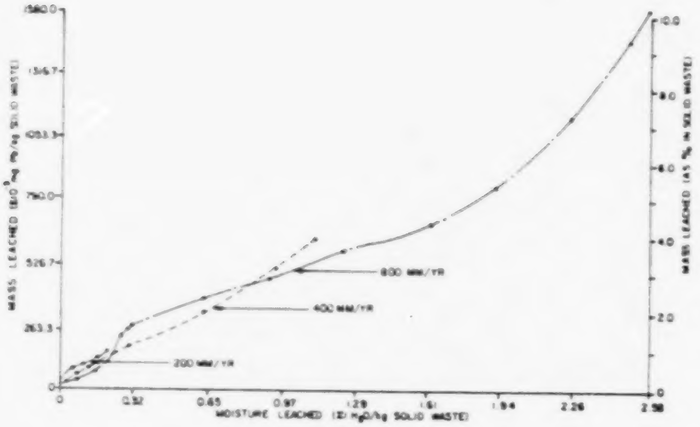

Figure 15. Mass Leached: Pb

To date, the 800 mm/year cell has released 10.5% of available lead. Because of the non-declining growth pattern evident, the ultimate release of lead is expected to be far higher.

GAS EVALUATION

Gas samples are extracted from all test cells on a monthly basis for compositional analysis. In addition, gas quantity data is recorded for interior Test Cells 16, 17, 18 and 19 on a daily basis. Because both quality and quantity data is available only for the interior cells, our attention here will be redirected to one of the interior municipal solid waste only cells, Test Cell 16.

A plot of gas composition with time for Test Cell 16 has been included as Figure 17. As shown, compositional data is available for 3½ years over a period stretching from ½ to 4 years after cell construction. Although methane composition reached an early peak of 20%, methane has generally been confined to less than 10% of the assayed gas over the first 2½ years. Since that time, methane has increased to approximately the 50% level, where it appears to stabilize at the fourth year. Carbon dioxide, on the other hand, appears to reach its highest composition levels early. A peak reading of 80% is noted only 3/4 of a year after cell construction. After this peak, carbon dioxide levels generally have declined to levels from 40% to

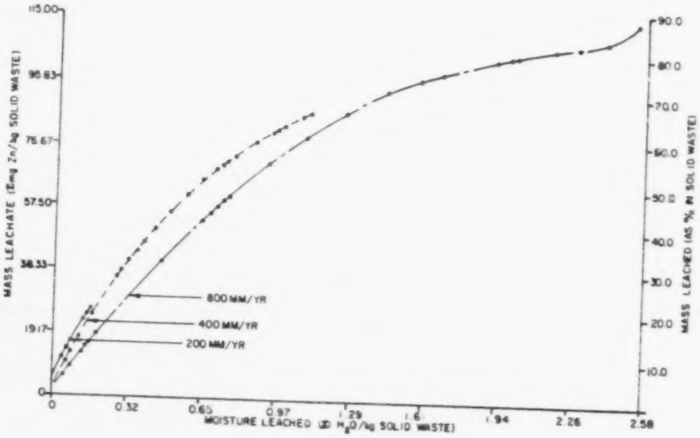

Figure 16. Mass Leached: Zn

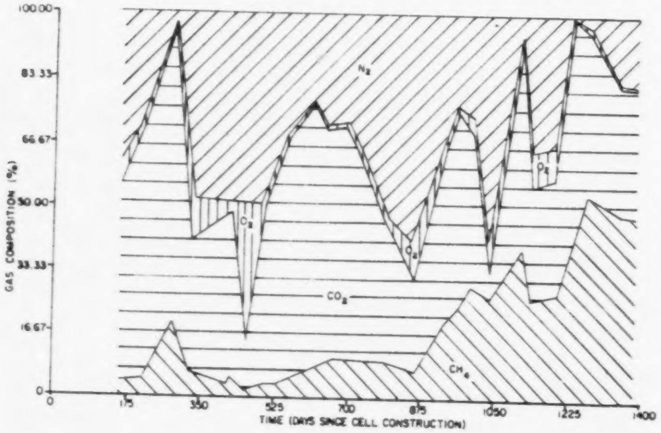

Figure 17. Gas Composition At Test Cell 16

50% through the fourth year. Typical atmospheric gases (nitrogen and oxygen) consume the balance of gas compositions and were found to contribute up to 85% of gas through the first year before declining steadily to levels between 0 and 20%. Other gases (e.g. hydrogen) were also detected over the time period shown but were at trace concentrations (less than 1%) and therefore, not shown. Generally, gas composition fluctuated dramatically with time. Despite these sometimes severe fluctuations, the overall trends shown in Figure 17 generally follow the theoretical gas production patterns reported by Rovers and Farquhar.(6)

A plot of gas quantity with time for Test Cell 16 has been included as Figure 18. Data included is for a 1-year period stretching from 2½ to 3½ years after test cell construction. As shown, gas volumes have fluctuated between 0.05 and 1.0 ml/kg/day. Overall trends indicate an average of approximately 0.5 ml/kg/day over this 1-year period for a cumulative gas generation of 0.18 l/kg/year. Because attempts to seal the test cells and quantify gas volumes were unsuccessful in the early years of this project, ultimate gas production volumes cannot be determined.

ACKNOWLEDGEMENTS

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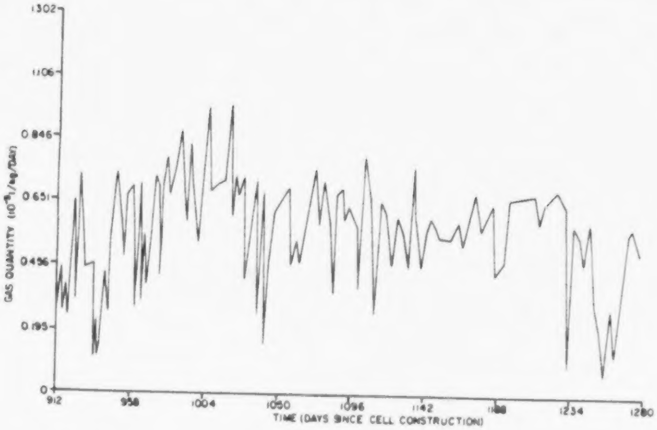

Figure 18. Gas Quantity At Test Cell 16

Protection Agency. The authors would like to express their appreciation to Mr. Dirk R. Brunner, Project Officer, for his assistance. The data base used in preparing this paper was obtained from the efforts of Systems Technology, Inc., pursuant to EPA Contract No. 68-03-2120.

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GAS PRODUCTION IN SANITARY LANDFILL SIMULATORS

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ABSTRACT

Gas compositions and production rates from sanitary landfill simulators during the initial 100 days of study were determined using conventional gas chromatography and a newly developed, automated gas production monitoring system. The automated gas monitoring system allows the simulator to release gas only when the presssure in the simulators exceeds surrounding atmospheric pressure by 2 cm water column. Measurement of gas flow with a thermoelectric device is an improvement over previous systems because the units do not have to be corrected for ambient temperature and pressure variation. This measuring system also eliminates leakage (through liquids and/or gaskets) which has complicated volumetric determinations in previous published studies. Gas flow, barometric pressure, ambient temperature, and test cell (simulator) temperature are automatically digitized and recorded by a data logger at hourly intervals. Gas samples were analyzed for oxygen, nitrogen, carbon dioxide, methane, hydrogen and water vapor. The gas analysis system and the production monitoring system are described and data for the first one hundred days of operation are presented.

Gas production measurements are in good agreement with those reported in the literature for similar systems. No methane or hydrogen has been detected in gas samples. Carbon dioxide has been the only product gas observed to date.

INTRODUCTION

Gas generation in sanitary landfills creates two major problems. Carbon dioxide produced by the fill goes into solution in surrounding groundwater and causes a "hardness halo" in the groundwater near the fill. Methane produced by anaerobic decomposition can migrate off-site and create hazardous conditions in surrounding structures. Many investigators have reported production of combustible gas from municipal solid wastes (1, 2, 3, 4, 5, 6), and economic methane recovery from sanitary landfills has been demonstrated (7, 8). Accurate measurement of the composition and amount of gas produced during decomposition of municipal solid wastes (MSW) is essential to both understanding the pollution problem and exploiting a possible fuel source.

Measurements of gas from decay of MSW in sealed containers (1, 2, 3, 5, 9, 10) have varied widely depending on the experimental conditions under which the observations were made. Experimental parameters which have affected gas production in MSW landfill simulators include moisture content, type of MSW (baled, shredded, unprocessed, or synthetic) degree of compaction, test cell size, and duration of test. Dewalle, Chian, and Hammerberg (1, 11) recently published a extensive review of the effects of these experimental variables on gas production from MSW and concluded that moisture and type of MSW were the major controlling parameters. The wide variability in published results from previous experiments indicated the need for a long-term, well-instrumented landfill simulator experiment.

The work described in this paper is intended to corroborate and add to previous determinations of gas production and gas composition from MSW using an improved gas monitoring methodology. Most investigators have relied on manometers (10), wet test meters (1, 5), and Mariotte flasks (1). One design of a sanitary landfill simulator proposed by Miller and others (12) incorporated a linear mass flowmeter with automatic flow integration and flow rate recording. The gas monitoring project described here parallels the previous work (1, 5, 6); but with more complete instrumentation. Two sizes of lysimeters, that are comparable in capacity to those used by other researchers were included in this investigation to allow a broad comparison of data obtained in this project with that obtained in other experimental systems. This paper describes a completely automated gas monitoring system, and sanitary landfill simulators (lysimeters or test cells) on which the system has been installed; reviews the characteristics of the system's performance to date and the first 100 days' data; and presents a comparison of the results obtained to published data from similar systems.

MATERIALS AND METHODS

Lysimeter Construction

The lysimeters or simulators employed in this study are cylindrical, steel (6.35 mm rolled plate) tanks with a coal-tar epoxy coating on all interior surfaces. The acid- and base-resistant, coal-tar epoxy protects the lysimeter walls from corrosion and the lysimeter contents from contamination. Two lysimeters (Numbers 10 and 22) have dimensions of 0.91 m inside diameter and 1.83 m height, and two (Numbers 15 and 16) have dimensions of 1.83 m inside diameter and 3.66 m height. In the former, 0.91 x 1.22 m cells of compacted MSW were placed on a soil layer that rests on a layer of polypropylene beads, and in the latter 1.83 x 2.44 m cells of compacted MSW were placed on a soil layer that rests on a layer of polypropylene beads (Figure 1). Unbaled, unshredded MSW was placed in 30.5 cm lifts and compacted to approximately 400 kg (wet wt)/cu m density in the larger lysimeters; smaller, randomly sized lifts were compacted to an approximate density of 405 kg (wet wt)/cu m in the smaller lysimeters. The weights of MSW placed in each lysimeter are given in Table 1. All MSW was obtained from non-commercial collection routes in Warren County, MS, in April, 1978.

A 500-kg composite sample of MSW was sorted at the time the lysimeters were filled and the resulting percent composition of the MSW used in the study is given in Table 2. Table 2 also presents composition data from other lysimeter studies.

After loading was completed on April 16, 1978, the lysimeters were sealed by welding steel lids to the tops of the lysimeters and by pumping metal sealant (3M Weatherban 202) into a machined groove (Figure 2) on the undersides of the lids. The details of the loading procedure for these four MSW lysimeters, the provisions for rainfall simulation and leachate collection and the temperature and barometric pressure instrumentation are described in a construction report (15).

Figure 3 is a photograph of the facility in which the lysimeters are housed. The lysimeters stand in a pit, one end of which can be seen in Figure 3. The pit, 1.5 m deep, allows access to the top of the tanks for servicing. The building is temperature-controlled at 25 ± 3°C.

Each lysimeter profile was constructed to resemble a core of MSW taken from a municipal landfill. The lysimeters simulate medium density cells in a sanitary landfill in the humid eastern United States containing unprocessed MSW. Infiltration is simulated by applying 66 cm of deionized water per year (at a rate of 1.27 cm per week). The initial moisture content of the MSW was 21.92% (of the dry weight).

Gas Flow Measuring and Sampling System

The gas flow measuring and sampling system consists of gas probes in each tank that collect gas produced by the decaying waste, a gas flow measuring and logging device and a system that allows the extraction of an uncontaminated gas sample for analysis. Figure 4 is a schematic diagram of the system.

Gas Probes

Three gas probes were installed in each lysimeter. Their locations within the municipal solid waste profile for a large lysimeter are shown in Figure 5. Comparable locations were used in the small lysimeters. Each gas probe (Figure 6) consists of a perforated 0.6 cm copper tube coated both inside and out with coal-

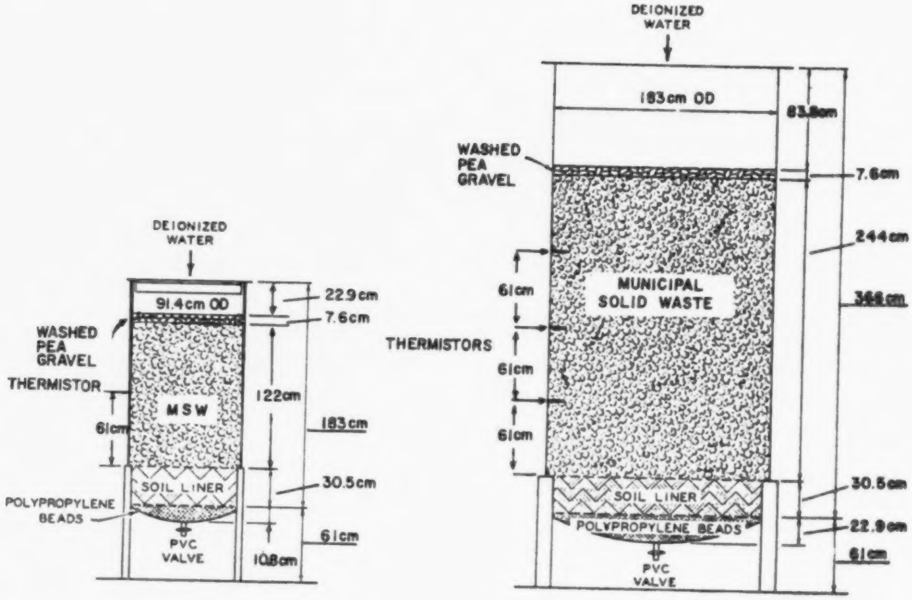
DEIONIZED WATER
91.4cm OD
22.9cm
7.6cm
WASHED PEA GRAVEL
THERMISTOR
MSW
61cm
122cm
183cm
SOIL LINER
30.5cm
POLYPROPYLENE BEADS
PVC VALVE
61cm
10.8cm
DEIONIZED WATER
183cm OD
63.8cm
WASHED PEA GRAVEL
7.6cm
244cm
MUNICIPAL SOLID WASTE
61cm
THERMISTORS
61cm
61cm
366cm
SOIL LINER
30.5cm
POLYPROPYLENE BEADS
22.9cm
61cm
PVC VALVE

Figure 1. Schematic diagram of the municipal solid waste lysimeters

TABLE 1. WEIGHTS OF MSW PLACED IN THE LYSIMETERS

Lysimeter No.	Wet Wt. (kg)	Dry Wt. (kg)
10	322	264
15	2555	2096
16	2555	2096
22	318	261

TABLE 2. PERCENT COMPOSITION OF MUNICIPAL SOLID WASTE USED IN THIS AND OTHER INVESTIGATIONS

Category	This Study	Jackson & Streng(13)	Chian and others(1)	Eifert & Swartzbaugh(14)
Paper	44.79*	40.53	36.5	49.6
Metal	10.82	8.29	14.7	9.5
Plastics, rubber, leather	9.03	6.52	2.8	6.0
Glass	7.61	7.42	6.8	12.0
Textiles	3.08	4.19	0.7	3.2
Disposable diapers	2.68	1.78	---	1.4
Food waste	0.94	7.53	14.4	7.3
Wood	0.49	0.86		
Garden waste	0.41	15.32	3.1	4.6
Ash, rock, dirt, fines	20.15	5.48	14.9	5.4

* All percentages are on a dry wt. basis.

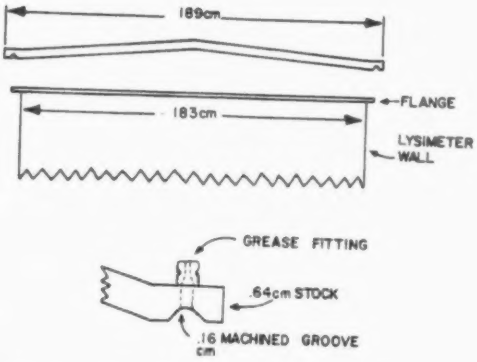
189cm
183cm
FLANGE
LYSIMETER WALL
GREASE FITTING
.64cm STOCK
.16 cm MACHINED GROOVE

Figure 2. Detail showing method of sealing the tops of the lysimeters.

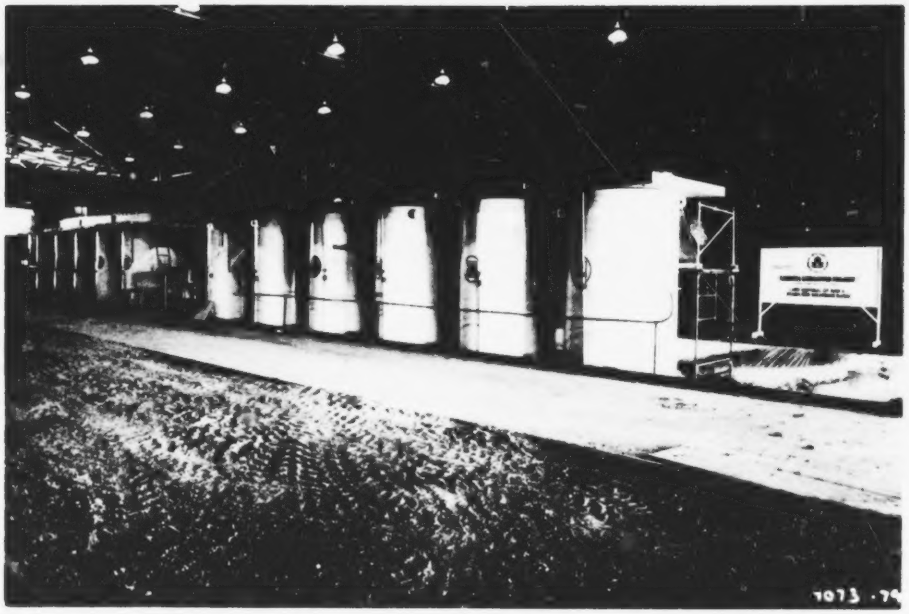

Figure 3. Facility for housing landfill simulator project.

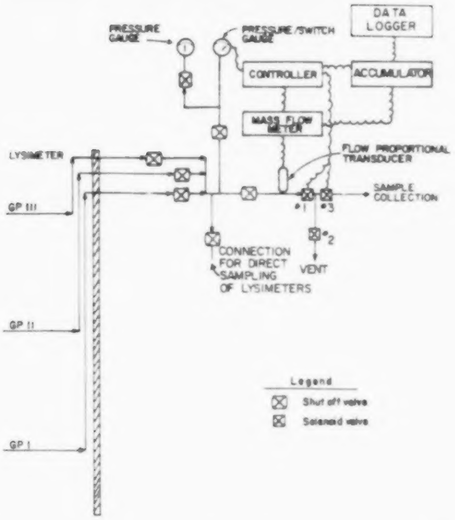
PRESSURE GAUGE
PRESSURE/SWITCH GAUGE
DATA LOGGER
CONTROLLER
ACCUMULATOR
MASS FLOW METER
FLOW PROPORTIONAL TRANSDUCER
LYSIMETER
GP III
GP II
GP I
SAMPLE COLLECTION
#1
#3
#2
VENT
CONNECTION FOR DIRECT SAMPLING OF LYSIMETERS
Legend
Shut off valve
Solenoid valve

Figure 4. Schematic diagram of gas flow measurement and sampling system. GP = Gas Collection Probe

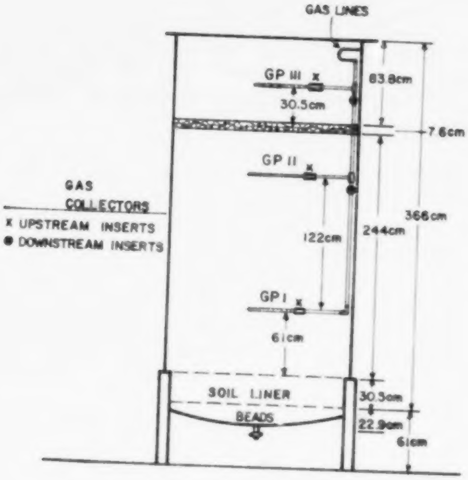
GAS LINES
GP III
30.5cm
83.8cm
7.6cm
GP II
GAS
COLLECTORS
X UPSTREAM INSERTS
● DOWNSTREAM INSERTS
122cm
244cm
366cm
GP I
61cm
SOIL LINER
BEADS
30.5cm
22.9cm
61cm

Figure 5. Position of gas collection probes in the large lysimeters

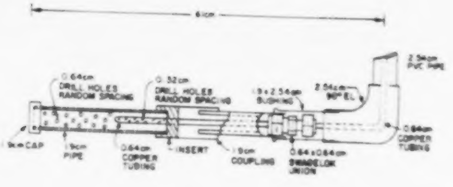
61cm
2.54cm PVC PIPE
0.64cm DRILL HOLES RANDOM SPACING
0.32cm DRILL HOLES RANDOM SPACING
1.9 x 2.54cm BUSHING
2.54cm 90° EL
0.64cm COPPER TUBING
1.9cm CAP
1.9cm PIPE
0.64cm COPPER TUBING
INSERT
1.9cm COUPLING
0.64 x 0.64cm SWAGELOK UNION

Figure 6. Detail of construction of gas collection probes

tar epoxy to prevent corrosion by leachate. The copper tubing was fitted into a perforated 1.9 cm PVC pipe to protect the tubing during compaction of the waste. Each gas probe was connected to a gas collection line using a union after the underlying municipal solid waste had been placed and compacted. An epoxy-sealed rubber insert in the PVC pipe upstream from the union prevents leachate contact with and corrosion of the union and attached copper gas lines. A 1.9 cm PVC pipe attached to the inside wall of the steel tank covers the gas lines as they run up the wall and out of the solid waste. The copper gas collection lines pass through the tank wall at a point just below the top of the lysimeter. At the exits, the lines were brazed to the exterior of the lysimeters. The PVC pipe and copper gas lines extending up the insides of the cells were attached in such a way so as to allow for compaction and consolidation of the municipal solid waste during the course of the experiment.

Gas Flow Measuring System

The gas flow measuring system operates on a pressure-controlled flow principle. Internal cell pressure is maintained at a pressure slightly above atmospheric by controlled out-gassing of the lysimeters. Gas flow is measured by a linear mass flowmeter. The internal tank pressure is normally allowed to reach a pressure of 4 cm water column (WC), with respect to external air pressure; then gas is bled off through the gas flowmeter until pressure is reduced to a low set point (usually 2 cm WC). The components of the gas flow measuring system for each cell are as follows: one differential pressure switch/gauge, three solenoid valves (normally closed), one linear mass gas flowmeter, one controller, seven shut-off valves, and one pressure gauge. A data logger (Martek DLS) records gas production data from all four lysimeters.

The differential pressure switch/gauge incorporates gauge and switch set point indicators for continuous indication of internal pressure and switch settings. These units are diaphragm-operated with switching accomplished by photocell-controlled relays. Set points are adjustable over a range of 0-5 cm WC. The unit can be regulated to 0.1 cm WC. The pressure switch/gauge is used to operate two normally-closed solenoid valves. When the pressure reaches the high set point (4 cm WC) the two solenoid valves open and the gas from the lysimeter passes through the flowmeter. When the pressure drops to the low set point (2 cm WC) the solenoid valves close. A pressure gauge (no switching capability) has been installed so that periodic checks can be made on the pressure switch/gauge's performance. The pressure switch/gauge is a Dywer Photohelic Model 3000-5-CM and the auxillary pressure gauge is a Dywer Magnehelic Model 2015.

The linear mass flowmeter consists of an electrically-heated tube and an arrangement of thermocouples which measure differential cooling caused by the gas passing through the tube. Thermoelectric elements generate DC voltages which are directly proportional to the rate of mass flow of gas through the tube. This proportionality permits the linear mass flowmeter calibrated for one gas to be used to measure the flow of a mixture of gases, such as landfill gas, if the gas composition and the thermal conductivity of the component gases are known. The mass flowmeter signal is insensitive to ambient pressure and temperature changes. These mass flowmeters because of their linear signal are ideal for use with totalizers and automatic data reduction equipment. The flowmeter, at its maximum flow rate (1 liter/min), generates a 5 VDC output. The flowmeters are Matheson Model 8116-0113 with Model H-1K transducers. The manufacturer's calibration was checked by comparing the recorded flow through each flowmeter with a reference flow standard (RFS). Where units varied more than ±0.5% from the RFS, they were recalibrated according to the manufacturer's instruction.

Three normally-closed solenoid valves downstream of the linear mass flowmeter open and close the gas lines. The valves are energized by a controller accepting input from the pressure switch/gauge and the mass flowmeter. Additional manually-operated shut-off and switching valves can be used to selectively sample different locations in the lysimeters and to close off the gas flow from the lysimeters in the event maintenance is required.

The controller, a solid state logic box, operates the solenoid valves as follows:

(1) When pressure reaches the high set point on the pressure switch/gauge, solenoid valves Nos. 1 and 2 (Figure 4) open and gas moves out of the lysimeter through the flowmeter.

(2) When and if the flow rate reaches 95% of the measuring capacity of the flowmeter, solenoid No. 2 closes to keep the gas flow rate in a measurable range.

(3) When the flow rate drops to 20% of the capacity of the flowmeter, solenoid No. 2 reopens. Steps (2) and (3) prevent gas flow rates from overranging the flowmeter.

(4) Solenoids 1 and 2 are activated until the pressure drops to the low set point of the pressure/switch gauge; both solenoids then close.

Analog signals from the linear mass flowmeter are digitalized through a voltage-to-frequency conversion and then accumulated in an electronic counter. Totals from the electronic counters (totalizers) are recorded on cassette tape at hourly intervals by a 56-channel, Martek DLS data logger. The counter registers are returned to zero after each hourly update. With each update the data logger also records day, hour (from an internal clock), temperatures from three thermistor locations in the large lysimeters and from one thermistor location in each of the two small lysimeters (Figure 1), barometric pressure, ambient air temperature from two locations in the lysimeter facility, and reference voltages for the electronics associated with temperature measurement. All recorded data are identified according to day and hour.

Gas Sample Collection

Two different sampling methods were tested during the first 18 days of lysimeter operation, one system used Tedlar bags, the other did not require Tedlar bags. Initially gas samples were composited into Tedlar gas bags and sampled for analysis using glass gas bulbs filled using a diaphram pump. This initial sampling configuration is shown in Figure 7a. Gases were composited in the Tedlar bags using a flow proportional sampling system operated by the electronic flow controller. For every 90 milliliters of gas registered on the flow counters, the controller closed solenoid No. 2 (Figure 4) directing 10 milliliters through solenoid No. 3 into a Tedlar bag. Tedlar bags of 2-mil thickness and 48-liter capacity were used. The glass gas bulbs are cylinders with O-ring (neoprene) sealed glass plungers that act as valves on each end. Tygon tubing was used as gas line from the Tedlar bags to the glass

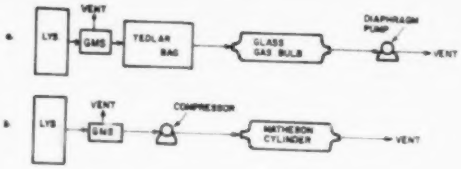

Figure 7. Diagram showing sampling systems used with MSW lysimeter gases.
System shown in 7a uses a Tedlar bag to composite sample.
System shown in 7b collects a "grab sample" through the gas monitoring system.

cylinders and from the glass cylinders to a diaphragm pump. The Tedlar bags were attached to the gas measuring system (GMS) so that the bags would be filled with gas composited over a given sample interval.

However, with this system, the expected oxygen depletion in the gas samples was not observed. The lysimeters including all fittings were checked for leaks with a gas leak detector (Gow-Mac Model 21-200) and no leaks were found. Excellent replication of gas analyses run after samples had been stored varying lengths of time (up to four hours) in the gas bulbs showed that the gas bulbs were not leaking. Gas samples were taken directly from a lysimeter at the branch in the GMS for direct sampling of the lysimeters (Figure 4) and at the same time from the Tedlar gas bag. The sample directly from the lysimeter was an order of magnitude lower in oxygen than the Tedlar bag sample. As a check for leaks, the Tedlar bag was filled with helium. Comparison of analyses of the bag's gas composition immediately after filling and 24 hours after filling showed a two-fold increase in oxygen content. Leakage could be due to permeation through the polymer material or to physical breaks in the bags. Suppliers caution that the Tedlar bags are very delicate and are easily damaged. By migrating through microvoids in the polymer's structure, gas molecules can permeate the Tedlar (16). Recently Polasek and Bullin (17) have reported that Tedlar bags were unsatisfactory for ambient air sampling. The combination of theoretical considerations and practical experience led to development of a system eliminating Tedlar bags.

As an alternative to the Tedlar bag composite sample, grab samples were collected from the lysimeters through the GMS in glass gas bulbs in the same bulb-pump procedure used before (Figure 7a) except that the Tedlar bag had been removed. Shortly after the change to grab samples had been made, the O-rings on the gas bulbs began to leak. To overcome these initial sampling problems an all-metal system using a compressor and steel storage tank was developed.

A small vacuum pump/compressor and a stainless steel gas cylinder is now used in the sampling configuration shown in Figure 7b. Copper tubing is used for gas lines from the GMS to the sample cylinder. Stainless steel quick-disconnects with double-end, shut-offs are employed throughout the system in order to prevent contamination during connection and disconnection of gas lines. The sample cylinder (Matheson, stainless steel, 150 mℓ capacity) is purged by pumping lysimeter gas through the cylinder. One end of the cylinder is then shut off, and lysimeter gas is pumped into the cylinder to a pressure of 2.9 kg/cm^2. The compressor is a KNF Neuberger, Inc., Model N04 STI diaphragm (Teflon) vacuum pump/compressor. This unit offers oil-free operation and contaminant-free pumping in a unit of relatively small size. Since the all-metal sample collection was initiated, the oxygen levels have been at or below detection limits (<0.1%) and the remaining components of the samples have been within expected ranges.

Gas Analysis

Gas samples are analyzed on a Perkin-Elmer Sigma 3 gas chromatograph (GC) for oxygen, hydrogen, nitrogen, carbon dioxide, methane and water vapor. The separation and quantification of these gases is performed on a 274 x 3.2 mm Carbosieve S column; the column temperature is programmed from 30°C to 175°C; helium carries the sample stream to a thermal conductivity detector. The GC is used with a Perkin-Elmer 056 recorder and an electronic peak integrator. The GC is calibrated with commercially prepared gas mixtures, and the results are reported in volume percent. The analytical technique used in this investigation does not separate oxygen and argon. Very low residual oxygen levels observed in the landfill gas are do to traces of argon in the gas samples.

DATA REDUCTION AND PRESENTATION

The system for data reduction and presentation is shown in Figure 8. The raw data tapes from the logger are read by a Martek Model 421-DRS magnetic tape reader and printed on a Anadex Model DP-500-9 paper tape printer. Another Martex Model 421-DRS magnetic tape reader is used to transfer data from the magnetic tape to a Hewlett-Packard Model 9830A computer through a Martek Model 421-12 DRS computer interfacing unit. The data fed from the Martek tapes to the HP 9830A is immediately printed on a Hewlett Packard Model 9866A printer at which time the data is verified by comparison with the printout from the Anadex unit. After verification, the data are reorganized and stored in raw form on magnetic tapes in an array that allows for easy access and manipulation by the computer. The tape files, each of which contains one update, are stored sequentially with the tape number and file number of the individual updates being recorded in a log book. As the raw data tapes become available, the raw data are reduced to engineering units by the computer using equations that correct for non-linear output from thermistors and barometric pressure sensors and for variation in gas composition. The reduced data for a whole day are stored on a new tape file along with the cummulative gas production for the day. In this manner, the computer builds a tape file library for each lysimeter containing all the reduced data for each day in a separate tape file to facilitate retrieval. The raw data tapes are preserved as originally generated.

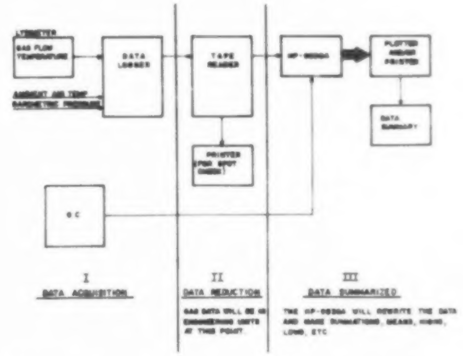

Figure 8. Diagram showing system for handling data from MSW lysimeters,

After storing the data on the proper tape file, the computer prints a data table for that day. The table contains the lysimeter number, the day; and for each update during the day, the time of the update and all the reduced data (gas production, temperatures, and barometric pressure).

A computer program was developed to plot the reduced data on a Hewlett-Packard Model 9862A plotter. The computer reads the data off of the reduced data tape file for the lysimeter of interest. The computer produces a plot of time versus gas production, temperature, and barometric pressure (all on the same plot) for a particular lysimeter and a particular day as specified by the user.

Another program was developed to plot daily cumulative gas production over a period of time. The computer inputs the daily cumulative gas production and the day from the reduced data tape file for the lysimeter of interest and plots production versus day for the period of time specified.

The computer can be programmed to perform other data manipulations as needed obtaining the data from the reduced data tape file for each lysimeter. The system is structured so that new data can be continually added while data reduction and manipulation continues. This system handles a large amount of data in a short time and at a very low cost. Such a data handling and storage task would be prohibitively expensive without the use of such a system.

RESULTS AND DISCUSSION

Quantity of Gas Produced

The total gas production for the landfill simulators for the initial 100 days is given in Table 3. The average value is 61.8 ml per kilogram (dry wt.) per day. The maximum value for gas production for the study is 69.5 mil per kilogram (dry wt) per day. Both the average and maximum rates are within the range reported for similar experiments (11, 2, 5). The production rates are higher than rates calculated for actual landfills (22-45 ml/kg/day)(11). It should be noted that these data are only

TABLE 3. SUMMARY OF GAS PRODUCTION FOR THE FIRST 100 DAYS

Lysimeter	Dry wt. MSW (kg)	Total Gas Production (liters)		Average Daily Gas Production (mℓ/kg (dw) · day)
10	264	1335 (1506)*		(57.0)
15	2096	13991		66.8
16	2096	14559		69.5
22	261	1401		53.8
			Average	61.8

* Failure of totalizer resulted in 10 days' loss of data. Value in parentheses is a projected gas production figure based on production before and after breakdown.

100 days' record and include the initial rapid CO_2 production associated with the beginning of decomposition.

A review of the literature indicates that a great deal of variability is to be expected in gas production data. Most systems would underestimate gas production due to leakage or equipment malfunction (11). There is also an inherent variability that is related to the composition of the refuse material in the simulator. Ramaswamy (9) demonstrated that the proportion of food scraps (or material added to act of food scraps) in the solid waste has significant effect of the volume of gas produced.

Figures 9 and 10 show two typical gas production curves over a 24-hour period for Lysimeter 15. These were selected to show how temperature and barometric pressure changes affect movement of gas out of a tank over the course of a day. The effect of temperature is especially obvious in Figure 10, where the gas flow can be shown to increase as the temperature increases in the building and in the headspace at the top of the simulator.

Figures 11-14 show gas production over a 100-day period for the lysimeters. The production fluctuates widely, reflecting changes in pressure differential caused by variations in temperature, barometric pressure, and microbial activity. The major feature in this data is an overall decline in gas production as the refuse stabilizes. Changes in production rate may occur when the lysimeters move into methane generation.

In general, the gas production rates reported here are within the range of rates observed by other researchers. The average rates are higher than the majority of landfill simulator gas production rates, perhaps reflecting an increased accuracy in the gas flow determination from the automated flow measuring system. The overall pattern of declining gas production matches the trend observed by other researchers; an initial gas surge that tapers off to a lower, more nearly constant production rate.

Gas Composition

Changes in the composition of the gas samples obtained from the lysimeters are shown in Figures 15-18. The oxygen level in all four lysimeters declined very rapidly after the tanks were sealed. All four of the tanks were sampled within 24 hours of sealing and the apparent oxygen (oxygen plus argon) had dropped to below 1.0%. The percent CO_2 shows a general increase in all four tanks. Nitrogen levels show an overall decline due to the sweeping of the tank by CO_2 generated by microbial metabolism. No hydrogen or methane was detected in any of the gas samples for the period reported here.

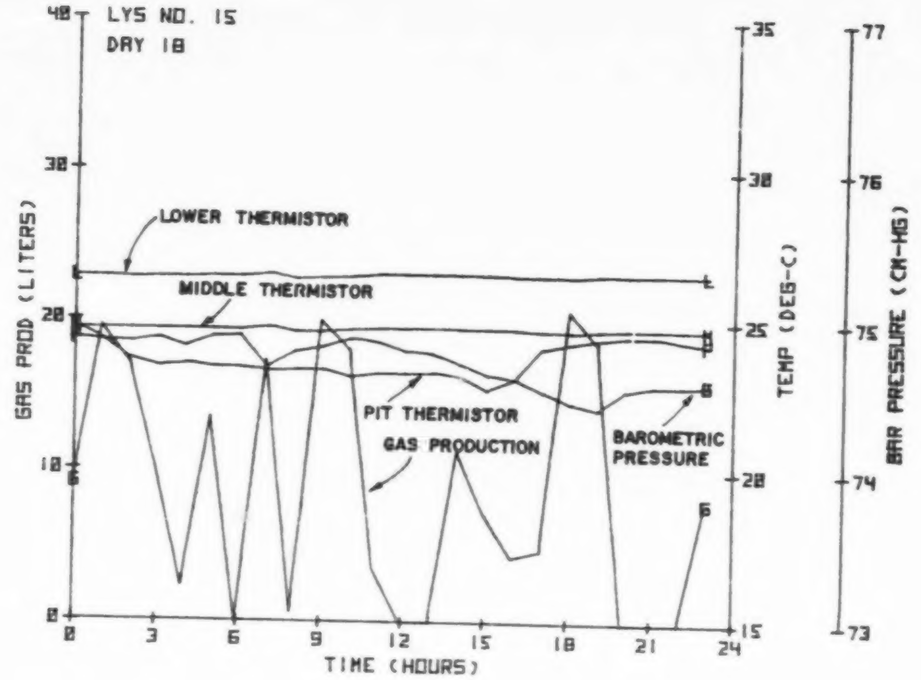

Figure 9. Plot showing variations in temperature, barometric pressure, and gas flow for day 18.

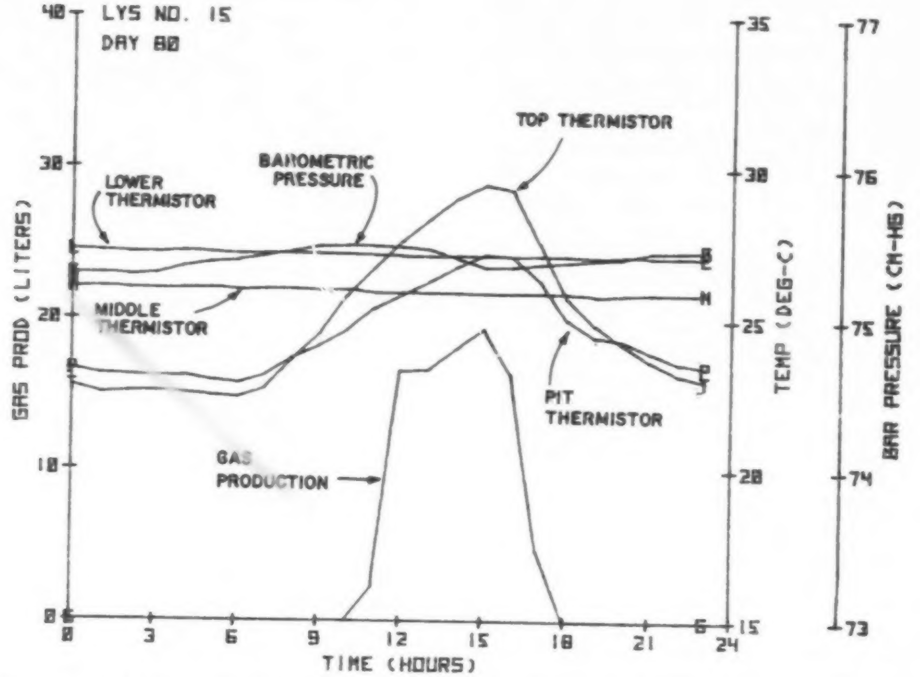

Figure 10. Plot showing variations in temperature, barometric pressure, and gas flow for day 80.

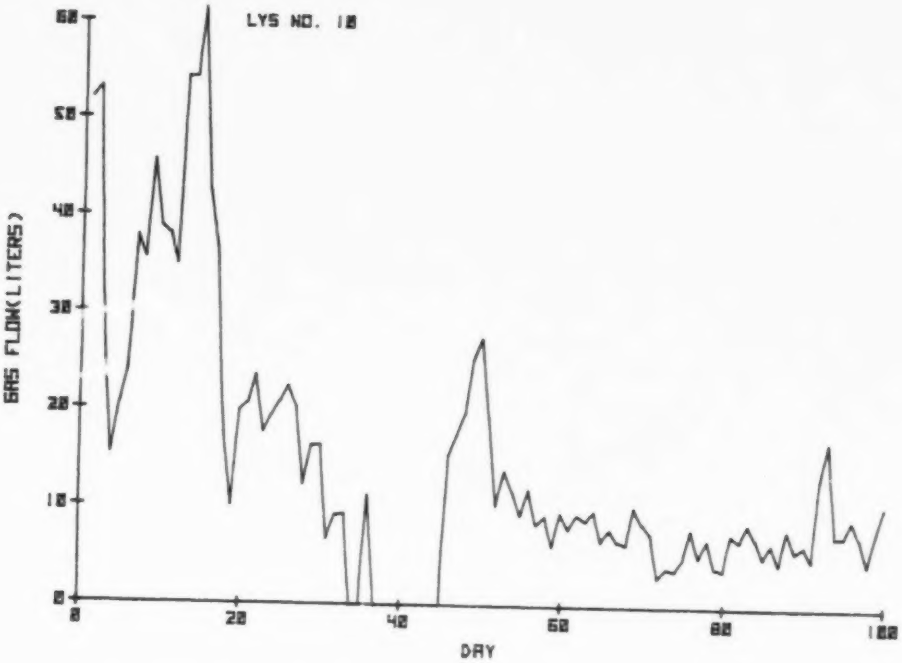

Figure 11. Plot showing variation in gas flow for lysimeter 10 for first 100 days.

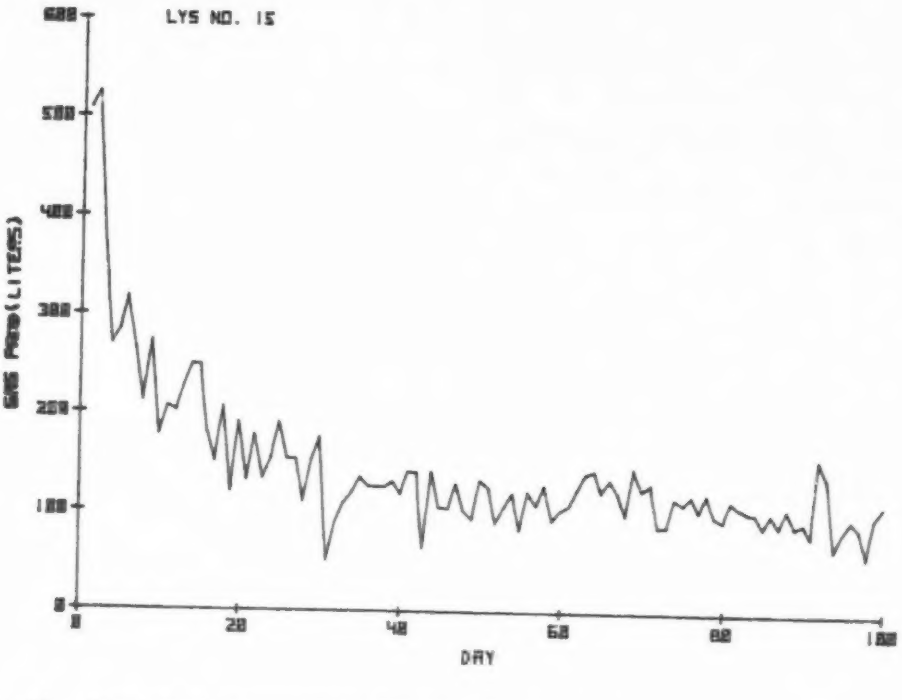

Figure 12. Plot showing variation in gas flow for lysimeter 15 for first 100 days.

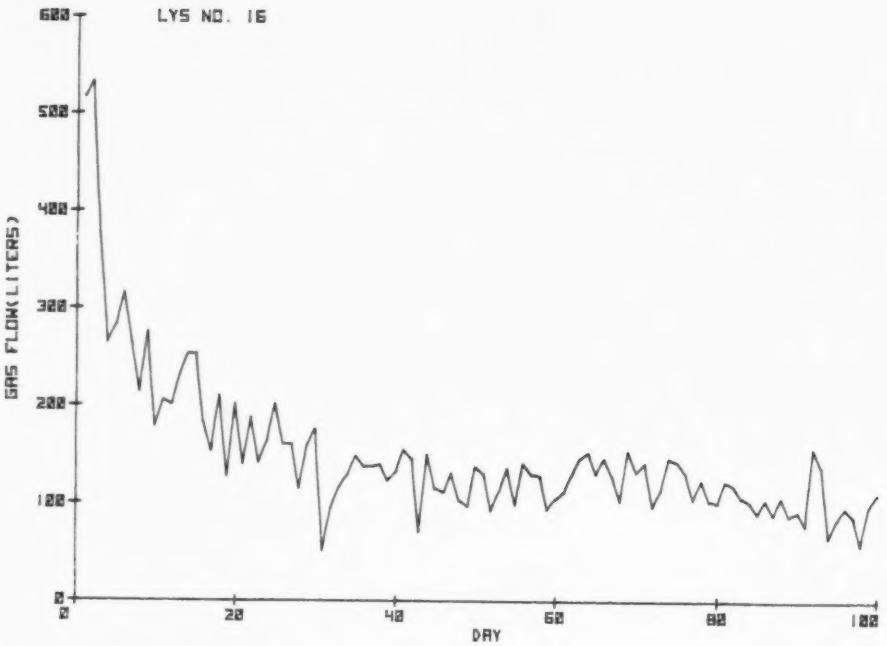

Figure 13. Plot showing variation in gas flow for lysimeter 16 for first 100 days.

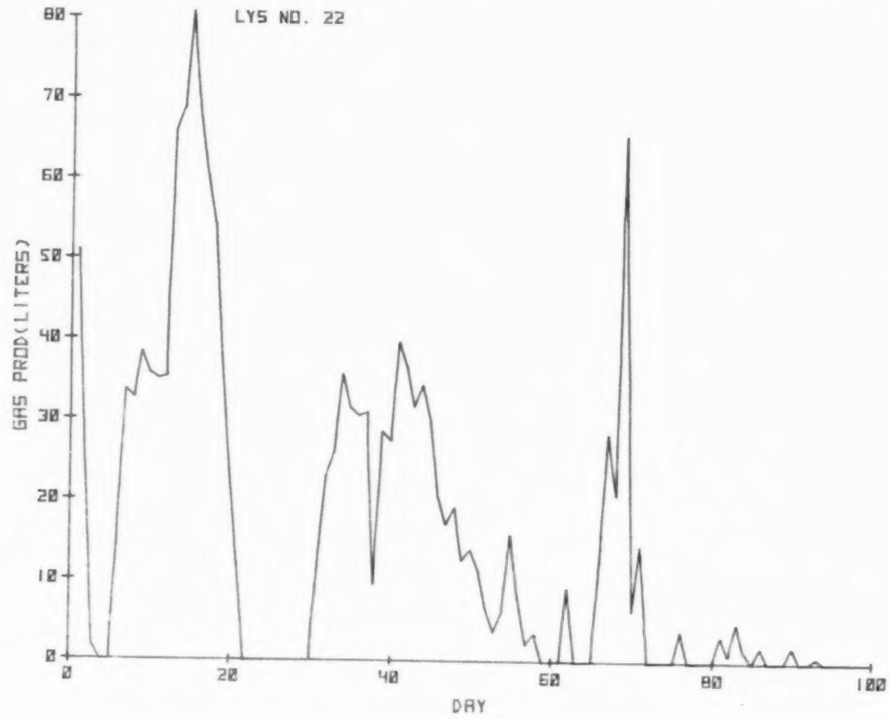

Figure 14. Plot showing variation in gas flow for lysimeter 22 for first 100 days.

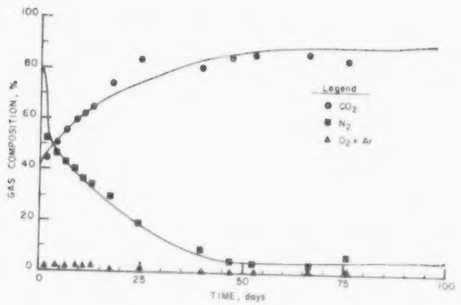

Figure 15. Plot showing variation in gas composition in lysimeter 10 for first 100 days.

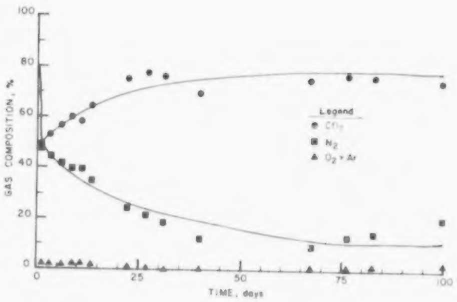

Figure 16. Plot showing variation in gas composition in lysimeter 15 for first 100 days.

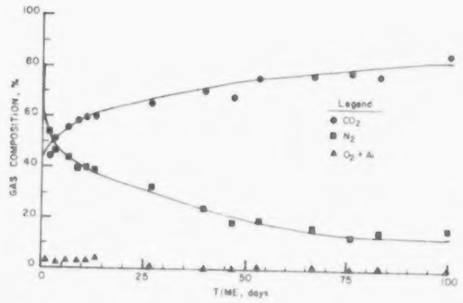

Figure 17. Plot showing variation in gas composition in lysimeter 16 for first 100 days.

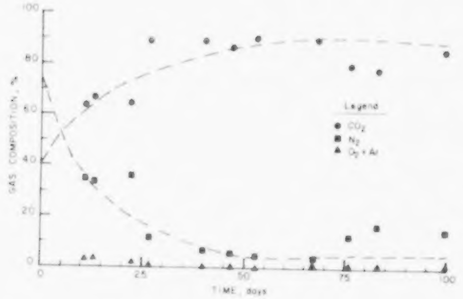

Figure 18. Plot showing variation in gas composition in lysimeter 22 for first 100 days.

The variation in gas composition observed in this investigation differs significantly from the ideal model proposed by Farquhar and Rovers (2). They proposed a gas production pattern with four distinct phases. The first phase involved nitrogen purging and oxygen depletion. The second phase involved increased hydrogen and carbon dioxide production. The third phase was characterized by a start-up of methane generation and a slight decrease and leveling-off of carbon dioxide evolution. The last phase involved development of a constant level of carbon dioxide and methane production. The aerobic decomposition in this study (Phase I) lasted only a very short time (<24 hours). The first measurements taken showed negligible oxygen levels. No hydrogen production (Phase II) or methane production (Phase III) was detected during the period reported here.

The results observed in the tanks used in this experiment approach closely those observed in other large tanks containing unshredded refuse. Merz and Stone (3) reported no significant gas production in large containers until 250 days into their experiment and no significant quantity of methane in the gas eventually generated. Rovers and Farquhar (10) observed a 50-day lag in gas production in the single sealed tank they used and no significant quantity of methane was recorded until 248 days had elapsed. Container sizes, temperature, moisture content, the character of refuse, degree of compaction, and the use of shredded as opposed to unshredded refuse, all appear to affect the composition of gas generated during stabilization. Other unmeasured or unmeasurable factors such as changing character of microbial populations and the availability of micronutrients probably affect the major stabilization processes and the types of decay gases produced at any particular time in any particular experimental setup.

While it is not possible to indicate why hydrogen and methane are absent in the gas samples coming from the landfill simulator, some obvious factors can be ruled out. Gas measurements have shown consistent, very low levels of apparent oxygen, indicating this should be no inhibitory factor. It has been proposed that water could carry oxygen into the lysimeters. Water has been added to the simulators at a rate equal to 1.27 cm of infiltration per week; but this small quantity of water could only contribute oxygen equal to 0.004% of the gas void available in the simulator. A small amount of O_2 introduced this way probably affects only the bacteria in the upper few centimeters of the simulator. Gas probes placed in the headspace of the tank, where any O_2 introduced should appear, showed no detectable differences in composition when compared to probes in the center or bottom of the tank.

The tanks have been maintained in a constant temperature facility within the range at which other experimenters have observed hydrogen and/or methane production. There is no indication of inhibitory temperature effects.

The refuse in the tank has not been shredded or compacted more than refuse placed in other experimental simulators. The excessive compaction and shredding have been indicated as inhibiting methane production (11).

The refuse in the tank has been increasing in moisture content. Simulated infiltration was provided at regular intervals so moisture fronts should have moved through the refuse, with successively more refuse becoming saturated. Thus, while the tank has not been totally saturated, parts of the refuse pile may have reached an optimum moisture content for methane production during the study. However, in-situ moisture measurements would be required to prove that suitable moisture regimes for methane production do exist.

As additional data on leachate pH, nutrient levels and organic substrates are collected, reasons for the particular gas composition observed in this study may become clearer. It appears that the simulators in this experiment will all reach field capacity before sigificant methane or hydrogen production is observed. Changes in the character of the leachate that can be correlated with the production of methane or hydrogen may provide information on what factors control methane or hydrogen generation in this experiment.

SUMMARY AND CONCLUSIONS

The first 100 days of data have been collected on gas production from four landfill simulators. An automated gas measuring system developed for the simulators is functioning satisfactorily. The system incorporates an electronic digital recording device that interfaces with a small computer to produce data listings and plots of data variation.

The average gas flow rates observed in the experiment are consistent with experimental values available in the literature. The average gas generation rates observed here are higher than field measurements of gas production at actual landfills. The pattern of declining gas production matches the trend observed by other researchers.

Gas composition is being determined at regular intervals using a gas chromatographic technique. An all-metal gas sampling system has been developed in order to obtain uncontaminated gas samples. Analyses available to date indicate that carbon dioxide is the product gas being generated in the lysimeters. These results are consistent with data available in the literature from several previous experiments.

ACKNOWLEDGEMENTS

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is the EPA Program Manager for this research.

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LEACHATE PRODUCTION FROM LANDFILLED MUNICIPAL WASTE

BOONE COUNTY FIELD SITE

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ABSTRACT

Five municipal waste test cells were constructed at the Boone County Field Site during 1971 and 1972. Two were field-scale and three were small-scale cells constructed for the purpose of performance comparison. All cells have been monitored for temperature, gas composition,settlement, and leachate quantity and characteristics. This paper presents interim results from the analysis of over four years of the leachate quantity and quality data. It must be emphasized that the results are from relatively shallow (less than 2.5 meters) batch-type cells.

INTRODUCTION

The Boone County Field Site consists of a ten acre tract located 8 km west of the City of Walton, Kentucky. Available facilities at the site include an office trailer, a pole barn, and a truck scale. An instrumentation shed has been placed near the test cell and a weather station has been erected at the site.

Elevations at the research site range between 213 and 244 m above sea level. Surficial soils at the site are predominantly a lean clay, classified by USDA as Nicholson silt loam. Rubbly limestone mixed with thin beds of soft calcareous shades of the Fairview formation underlie the shallow soil. The mean annual precipitation in the area is 927 mm. Monthly normal mean temperatures range from 0 deg. C in January to 24.4 deg. C. in July.

The objectives of test cells 1, 2A, 2B, 2C, and 2D included:

(1) Analyze the amount and characteristics of leachate.

(2) Analyze the composition of gases present in the cells.

(3) Evaluate a clayey-silt soil as an impervious liner for leachate control.

(4) Evaluate the behavior of a field-scale test cell, 2D, as compared to similarly constructed small-scale test cells, 2A, 2B, and 2C.

TEST CELL DESCRIPTION

Test Cell 1 consisted of a 45.4 m long by 9.2 m wide trench type sanitary landfill cell. The trench was excavated with vertical side walls and ramps on both ends sloping approximately 1:7. The center 15.3 m of the trench was sloped approximately 7 percent to the transverse center line as shown in Figure 1. A 30 mil (.76 mm) synthetic liner (Hypalon) was centered in the base of the cell to prevent any leachate from migrating below the cell. Directly above the liner a slotted collection pipe was installed along the transverse center line. Above this pipe and the synthetic liner a 45.7 cm thick clayey-silt soil liner was placed.

A second slotted collection pipe was installed in a trench in the soil liner. Short-circuiting to the lower pipe was prevented by lining the base and sides of the trench with a polyethylene strip. Both pipes drained into the observation unit beside the cell. The area around both pipes was backfilled with clean silica gravel. Details of the drainage scheme are shown in Figure 2.

During June 1971, 395 metric tons of residential refuse was placed in the test cell at a dry density of 429 kg/cubic meter. Moisture, temperature, gas and settlement probes were placed within the cell as it was constructed. Approximately .6 m of the compacted clayey-silt (CL) soil was placed over the refuse.

Four additional test cells containing municipal solid waste were constructed during August 1972. Three of the cells, 2A, 2B, and 2C were enclosed in identical cylindrical steel pipes, 1.83 m in diameter and 3.66 m long. The fourth test cell, 2D, was an 8.53 m square field-scale cell constructed to compare performance with the small-scale cells.

The steel pipes were coated with cold tar epoxy and placed vertically in an excavation on reinforced concrete pads. Slotted pipe in a trough in the concrete pad was used to collect leachate. Earth backfill was placed around the pipes to within 150 mm of the tops. Refuse was placed in the pipes in 90-135 Kg increments and compacted by dropping a 135 Kg weight from a height of approximately 1.2 m above the refuse. Temperature and gas probes were placed at several levels within the cell during refuse placement. 300 mm of compacted soil was placed above the refuse and then 300 mm of pea gravel to allow rapid percolation of rainfall and to minimize evaporation.

Cell 2D was constructed in an excavation 8.53 m square and 3.20 m deep. A 30 mil CPE liner was placed above a shaped sand bed in the base of the cell and extending up the sidewalls. A slotted PVC pipe was placed along the center line of the base of the cell for leachate collection and gravity drainage to the collection well. The entire base of the cell and liner was covered with 300 mm of silica sand. Plywood sheets were placed against the synthetic liner on the sidewalls for protection from puncture and tearing during cell filling. Construction details are shown in Figure 3.

A small bulldozer was lowered into the cell by a crane for compaction of refuse. Temperature and gas probes were placed during the filling at locations shown in Figure 3. A 300 mm layer of compacted soil cover was placed over the 2.44 m of refuse. A berm system, consisting of 150 mm high triangular-shaped clay berms, was hand constructed on top of the soil cover. This was constructed to promote uniform percolation of rainfall into the refuse cell.

Samples were obtained from the residential refuse being placed in all cells for moisture and composition studies. Chemical analysis of the refuse was done for a number of parameters for Test Cell 1. Statistical analysis indicated that any performance variations within cells 2A,2B and 2C could not be attributed to the differences in the composition or the original moisture. A summary of the cell data is shown in Table 1.

LEACHATE QUANTITY

TEST CELL 1

Leachate was initially collected from both the upper and lower pipes in Test Cell 1 approximately two months after construction. Cumulative leachate volume is shown in Figure 4 together with the volume of leachate predicted by using the water balance method. Leachate collected was 32% of the precipitation after 4.5 years.

In computing the water balance it was assumed that the monthly mean temperatures for the Boone County site were the same as that reported for Covington, Kentucky. The runoff co-efficients for the wet and dry seasons were chosen as .17 and .13. Assuming a soil moisture storage capacity of 125 mm for the final cover, 90 mm of water were required to bring the final soil cover to field capacity. An additional 106,000 liters of water were required to bring the entire cell and the soil liner to field capacity based on an estimated refuse field capacity of 330 mm/m of refuse.

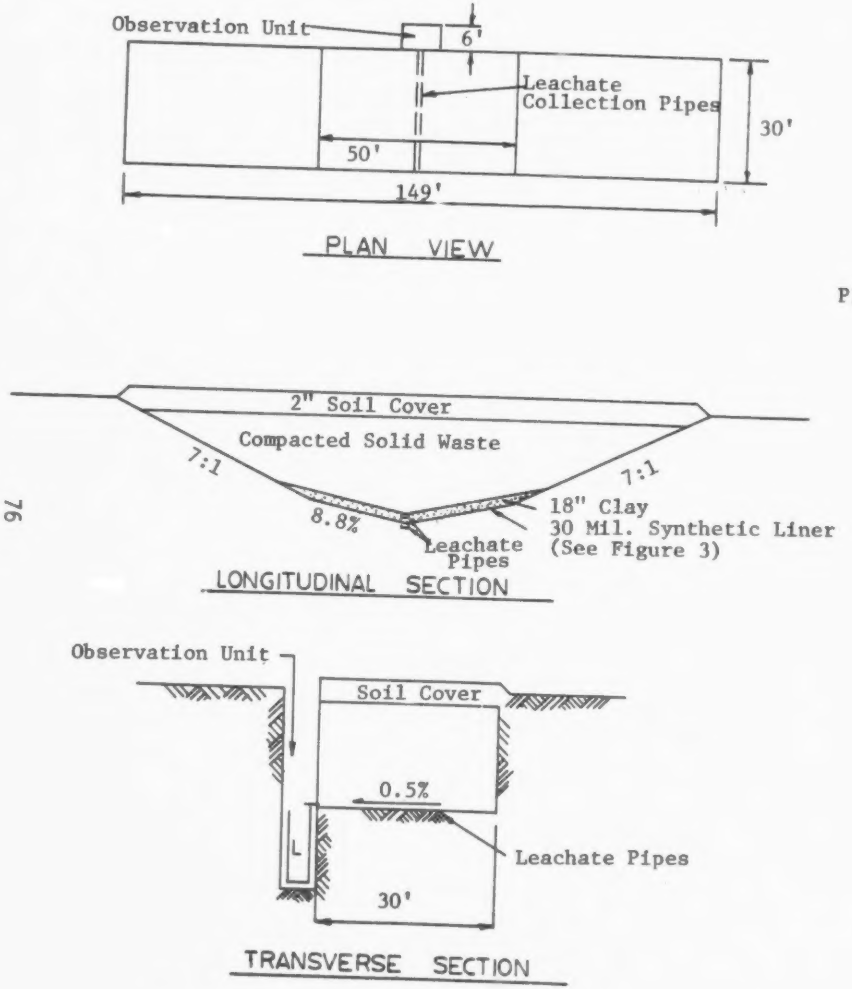
Observation Unit
6'
Leachate Collection Pipes
30'
50'
149'
PLAN VIEW
2" Soil Cover
Compacted Solid Waste
7:1
7:1
8.8%
18" Clay
30 Mil. Synthetic Liner
(See Figure 3)
Leachate Pipes
LONGITUDINAL SECTION
Observation Unit
Soil Cover
0.5%
L
Leachate Pipes
30'
TRANSVERSE SECTION

Figure 1. Design of Cell No. 1

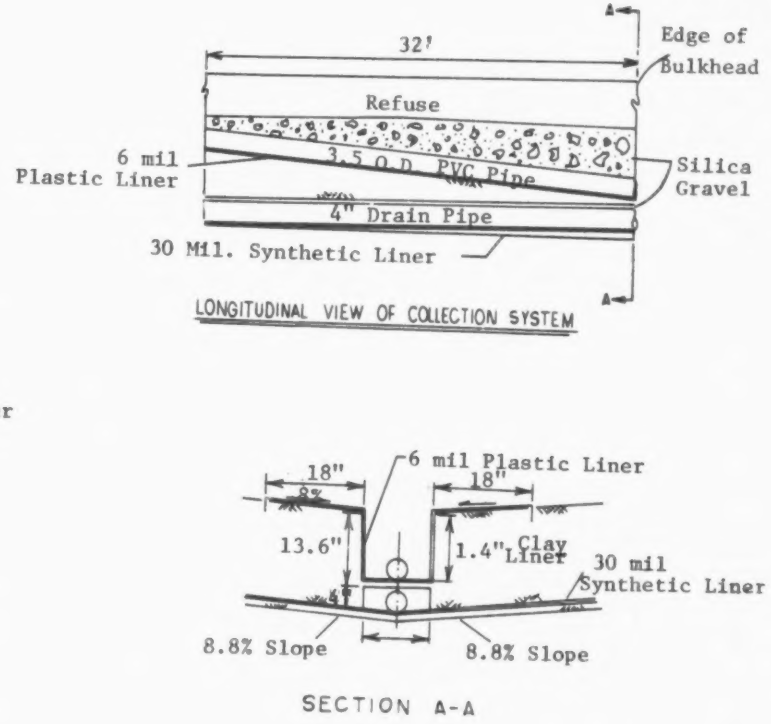
A
32'
Edge of Bulkhead
Refuse
6 mil Plastic Liner
3.5 O.D. PVC Pipe
Silica Gravel
4" Drain Pipe
30 Mil. Synthetic Liner
A
LONGITUDINAL VIEW OF COLLECTION SYSTEM
18"
6 mil Plastic Liner
18"
8%
13.6"
1.4"
Clay Liner
30 mil Synthetic Liner
8.8% Slope
8.8% Slope
SECTION A-A

Figure 2. Leachate Collection System.

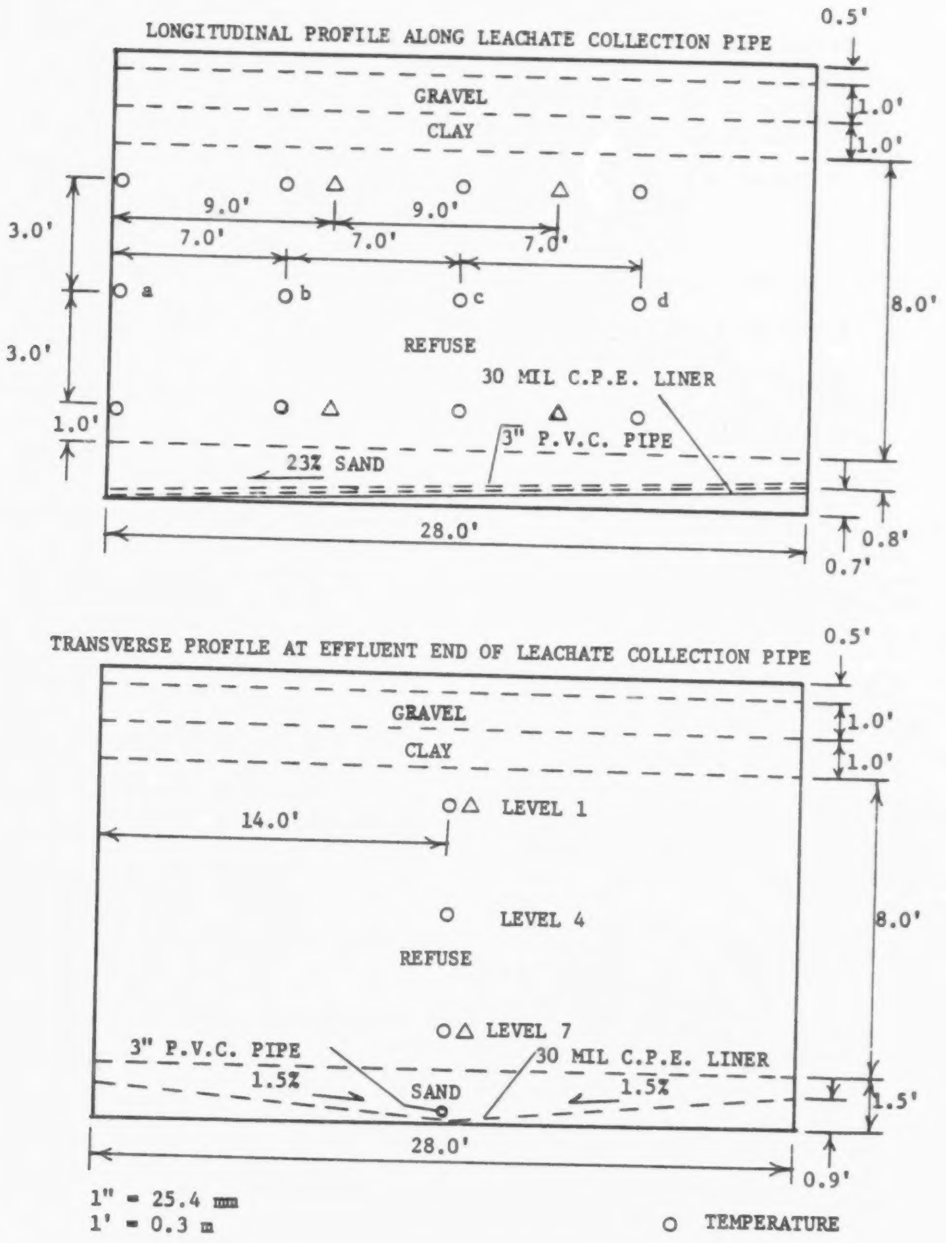
LONGITUDINAL PROFILE ALONG LEACHATE COLLECTION PIPE
0.5'
GRAVEL
CLAY
1.0'
1.0'
3.0'
9.0'
9.0'
7.0'
7.0'
7.0'
a
b
c
d
3.0'
REFUSE
8.0'
30 MIL C.P.E. LINER
1.0'
3" P.V.C. PIPE
2% SAND
28.0'
0.8'
0.7'
TRANSVERSE PROFILE AT EFFLUENT END OF LEACHATE COLLECTION PIPE
0.5'
GRAVEL
CLAY
1.0'
1.0'
14.0'
LEVEL 1
LEVEL 4
REFUSE
8.0'
LEVEL 7
3" P.V.C. PIPE
1.5%
SAND
30 MIL C.P.E. LINER
1.5%
1.5'
28.0'
0.9'
1" = 25.4 mm
1' = 0.3 m
○ TEMPERATURE
△ GAS

Figure 3. Construction details for cell 2D.

TABLE 1

SUMMARY OF CELL DATA

	Test Cell			
	2A	2B	2C	2D
Cover soil classification	CL	CL	CL	CL
Depth of soil cover, m	.30	.30	.30	.30
Surface area of refuse, m^2	2.627	2.627	2.627	72.83
Depth of refuse, m	2.56	2.56	2.56	2.44
Mass of refuse, kg (wet)	2640	2898	2814	106,231
Wet density of refuse, kg/m^3	392.6	430.9	418.4	597.8
Moisture content of refuse, % wet weight	22.5	27.1	24.1	31.8

The water balance calculations were reasonably accurate in predicting the quantity of leachate, with only a 12.7% difference after 4.5 years. If average evapotranspiration values had been used, rather than ones computed from the actual climatic conditions experienced, the difference would have been 35%. This large difference indicates a need for leachate volume calculations in designs to be based on extreme weather conditions as well as average values.

SOIL LINER

The quantity of leachate collected from both the upper and lower pipes and the amount of precipitation is listed in Table 2. The quantity of leachate from the lower pipe was equal to or greater than that volume from the upper pipe until January, 1972. This was caused by leaving the valve closed on the upper pipe except for weekly sampling, thereby inducing sufficient head to cause leakage into the lower pipe. After December 20, 1971, the upper pipe was allowed to drain freely through an S-trap into a sump in the observation unit, thereby greatly reducing the hydraulic head and consequently reducing the flow from the lower pipe.

The percentage for the leachate collected from the lower pipe has decreased over the years due to the change in collection method, and/or a decrease in permeability of the soil liner, to a point where more than 99% of the leachate is collected from the upper pipe. This indicates that the soil liner, maintained under a free flowing condition, is preventing virtually any of the leachate from migrating to the lower pipe.

From Table 2 a tendency can be noted that a greater percentage of the total flow comes from the lower pipe during the months of low total flow. For example, during 1974, 50% of the flow in the lower pipe occurred during six months when only 17% of the total flow occurred. Apparently during periods of high total flow, the soil liner becomes saturated, and with a higher permeability existing at the refuse-soil interface the leachate flows principally to the upper pipe. During periods of low total flow, the soil liner is not fully saturated, allowing percolation down to the liner and lower pipe in a larger percentage than when the soil is fully saturated. It is possible that the synthetic liner and the soil liner are functioning together and that the large quantity collected in the upper pipe is not due to the soil alone. It appears that the synthetic liner is responsible for at least some portion of the flow to the upper pipe by not allowing any deep percolation and by keeping the soil liner in a saturated condition for a longer time than if downward percolation were allowed.

TABLE 2. SEASONAL DISTRIBUTION OF COLLECTED LEACHATE

Months	Year	Upper Pipe[a]	Upper Pipe[b]	Lower Pipe[a]	Lower Pipe[b]
June-Aug.	1971	116	52	107	48
Sept.-Nov.	1971	135	36	243	64
Dec.-Feb.	1971-1972[c]	2,240	81	539	19
Mar.-May	1972	6,731	91	661	9
June-Aug.	1972	1,485	96	66	4
Sept.-Nov.	1972	16,580	95	880	5
Dec.-Feb.	1972-1973	62,096	99.4	395	0.6
Mar.-May	1973	127,430	99.9	156	0.1
June-Aug.	1973	57,536	99.7	175	0.3
Sept.-Nov.	1973	48,582	99.8	97	0.2
Dec.-Feb.	1973-1974	43,917	99.9	50	0.1
Mar.-May	1974	65,872	99.8	131	0.2
June-Aug.	1974	25,214	99.7	72	0.3
Sept.-Nov.	1974	45,829	99.8	79	0.2
Dec.-Feb.	1974-1975	90,123	99.9	51	0.1
Mar.-May	1975	80,792	99.9	55	0.1
June-Aug.	1975	34,960	99.7	106	0.3
Sept.-Nov.	1975	27,993	99.7	93	0.3

a. Volume in liters
b. Percent of total
c. Beginning of freely drained condition on January 1, 1972

TEST CELLS 2A, 2B, 2C, 2D

The experimental design called for the input of approximately 500 mm of precipitation each year into all of the cells. Average annual rainfall at the site is in excess of 900 mm so the cells were periodically covered, the cylinders with caps and 2D with nylon reinforced Hypalon.

Leachate was initially collected from each test cell on the date and at the cumulative rainfall quantities in Table 3. The early appearance of leachate from 2D, the field-scale cell, also occurred in Cell 1, indicating some short-circuiting.

Figure 5 shows the quantity of leachate collected from each test cell and with precipitation. Test Cell 2C produced very little leachate during the reporting period in comparison to 2A and 2B. A test boring in the cell did not show any free water stored in the cylinder. It was assumed that a leak developed at a welded joint near the surface of the soil cover and very little of the precipitation actually entered the refuse mass.

The quantities of leachate collected from 2A and 2B vary slightly, but by the end of 1976 50% more than 2A and 72% more than 2B per unit of surface area had been collected from 2D. One possible cause of this large difference could be leakage into Cell 2D through the walls. There also could be leakage through the cover. The quantity of leachate collected from 2D was in excess of the precipitation that occurred when the cover was removed.

The time delay between initial precipitation input and eventual steady leachate production results from the absorptive capacity of the refuse being achieved, or field capacity being reached. The field capacity for the refuse can be estimated from the work of Fungaroli and Steiner, knowing the in-place density of the refuse. The values are presented in the first column of Table 4.

The water required to bring the test cells to field capacity can be estimated using these values presented in the first column times the initial refuse depth, and then subtracting the initial moisture stored in the refuse and adding in 50 mm of water required to achieve field capacity in the cover soil. This value is listed in the second column of Table 4.

In order to compare this estimated value with the apparent water actually required, the apparent water required was chosen as that precipitation less leachate value when leachate production initially became steady. This apparent water requirement is in the third column of Table 4. The apparent field capacity is presented in the fourth column.

The values of estimated and apparent field capacity compare favorably, being within 10% or less of each other for all cells. Values might be closer, but the sequence of covering and uncovering the cells made leachate flow erratic and estimating the start of leachate flow was difficult.

LEACHATE QUALITY

Leachate samples were generally analyzed on a bi-weekly schedule. Work from test cell 1 indicated that concentrations were more total water flow than time dependent so the sample concentration data were reduced to weighted mean concentrations at the approximate time at which 100 mm intervals of leachate flow were recorded. This normalized the data for each cell so that concentration histories were based

TABLE 3. INITIAL COLLECTION OF LEACHATE

Test Cell	DATE LEACHATE COLLECTED	CUMULATIVE PRECIPITATION[a]
2A	6-15-73	724
2B	2-13-73	588
2C	6-19-73	765
2D	9-25-72	51

a. Millimeters

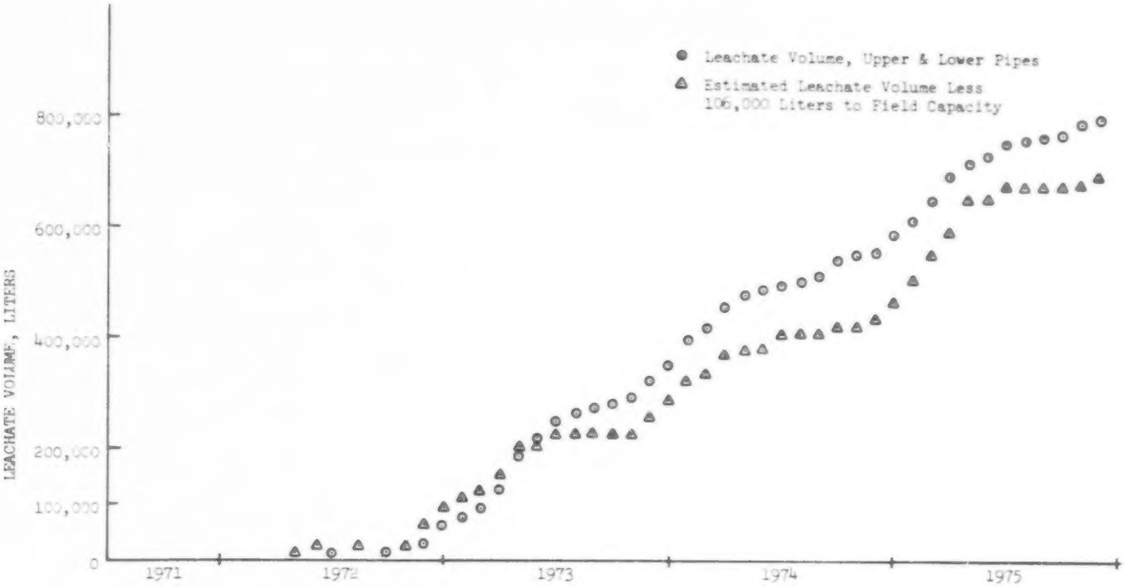

Figure 4. Cumulative leachate volume and estimated percolation.

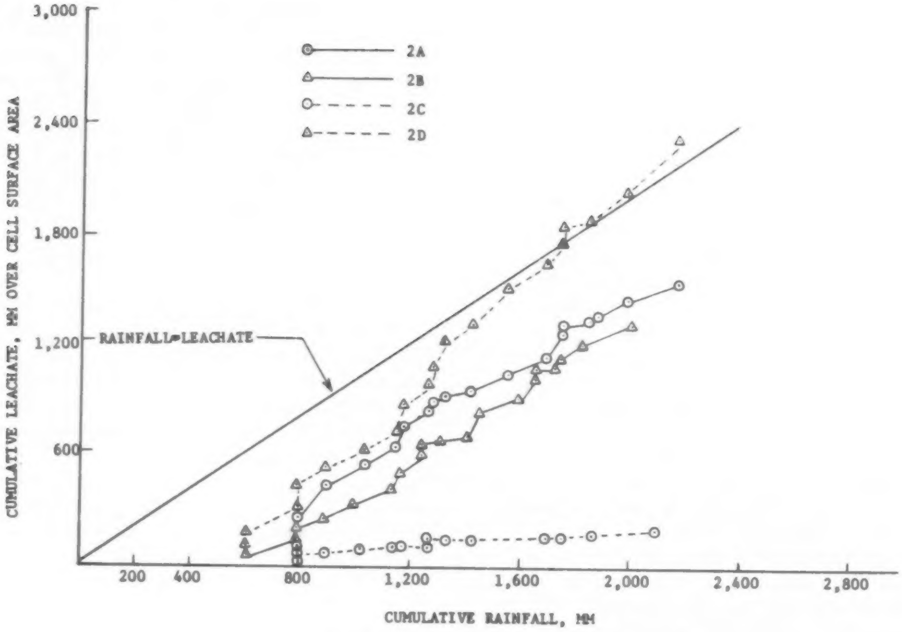

Figure 5. **Cumulative leachate collected.**

TABLE 4. REFUSE FIELD CAPACITY

Test Cell	Estimated Refuse Field Capacity[a]	Estimated Water Required to Reach Field Capacity[b]	Apparent Water Required to Reach Field Capacity[b]	Apparent Refuse Field Capacity[c]
2A	350	706	720	356
2B	358	669	590	327
2D	400	560	500	375

a. millimeters/meter of refuse depth, after Fungaroli and Steiner

b. millimeters

c. millimeters/meter of refuse depth

on a parameter accounting for the varying leachate quantities collected from each cell. Concentration and mass removal histories are presented in Figures 6-15.

A summary of peak concentrations and the date of the sample is presented in Table 5 for selected parameters. It is notable that all but one of the peak concentrations for test cell 2A occurred within a two month time span starting with the onset of leachate production. This was the time period during which or shortly after field capacity was achieved. Apparently these peak concentrations were the result of initial water contact with the refuse when the supply of the leachable substances and the contact time was high.

For test cell 2B the time span for peaks was somewhat longer than 2A, ranging to over 6 months. Test cell 2B did begin leachate production 4 months earlier than 2A and before the estimated field capacity was reached so this range might actually have coincided closely with 2A with the peak concentrations occurring during or shortly after field capacity was reached.

For test cell 2D the general time span for peaks was four months, with calcium and total hardness somewhat later. The volume of leachate collected prior to and during this time range was much greater than that for 2A and 2B and somewhat later than when sufficient precipitation had entered to satisfy both the apparent and estimated water requirements. It did not appear that the peak concentrations for 2D occurred during the period that field capacity was reached as occurred for 2A, 2B and test cell 1. If peaks did occur during this period, then the high concentrations in the leachate would had to have been either reduced by significant diluting leakage from the sides of the cell or by channelling through the cell. The latter situation would result in field capacity not actually having been achieved until somewhat after estimated water requirements had been met, possibly during the time and leachate volume range when peak concentrations were recorded. Dilution was indicated by the lower magnitude of almost all of the peak concentrations of 2D as compared to 2A and 2B.

Leachate Composition Comparibility

One of the primary objectives of the later test cells was to evaluate the behavior of a field-scale test cell, 2D, compared to similarly constructed small-scale test cells 2A, 2B and 2C. The concept was to determine whether similarity existed between individual small-scale cells as well as similarity between the small-scale cells and the field-scale cell, 2D. It was hoped that the small-scale cells would be adequate models of the large scale cell so that future research efforts might utilize the smaller cells for prediction of field behavior.

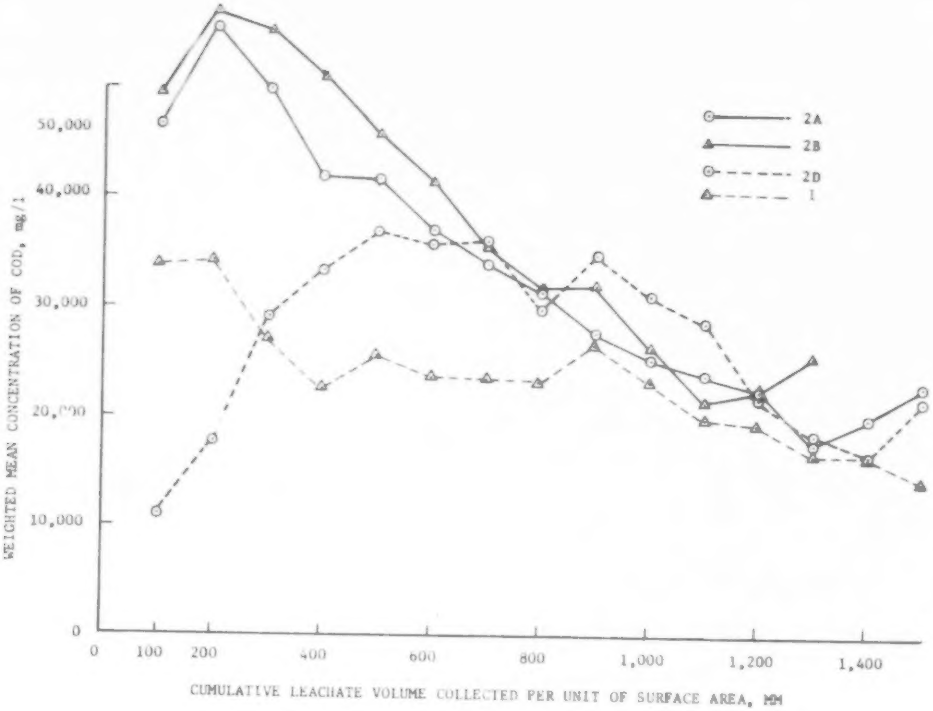

Figure 6 . Weighted mean COD concentration.

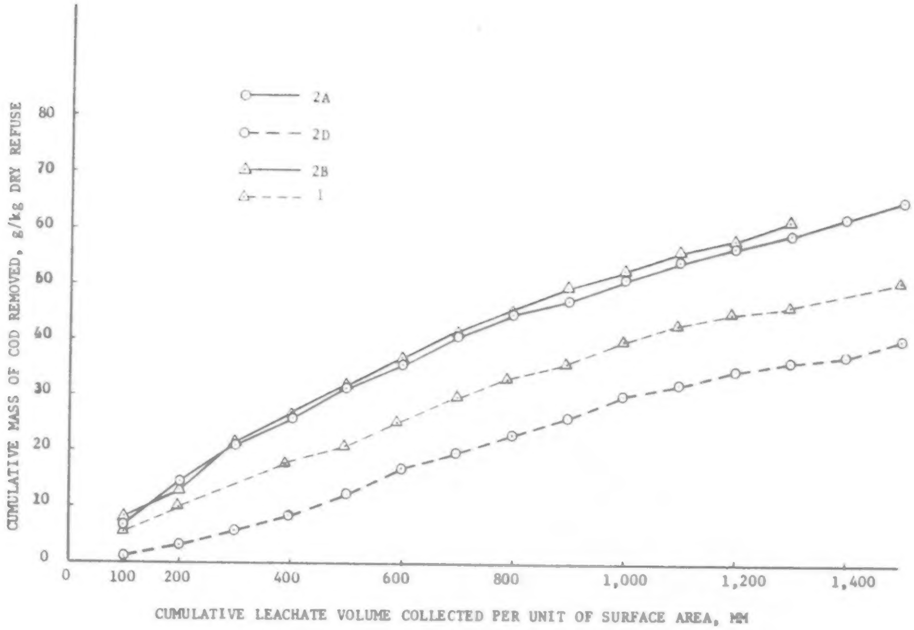

Figure 7. Cumulative COD mass removal.

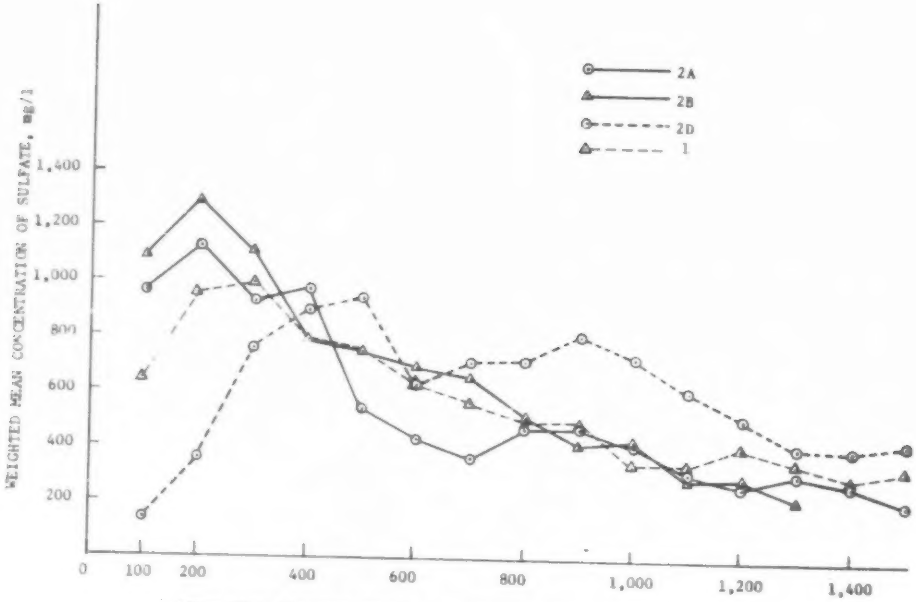

Figure 8 . Weighted mean sulfate concentration.

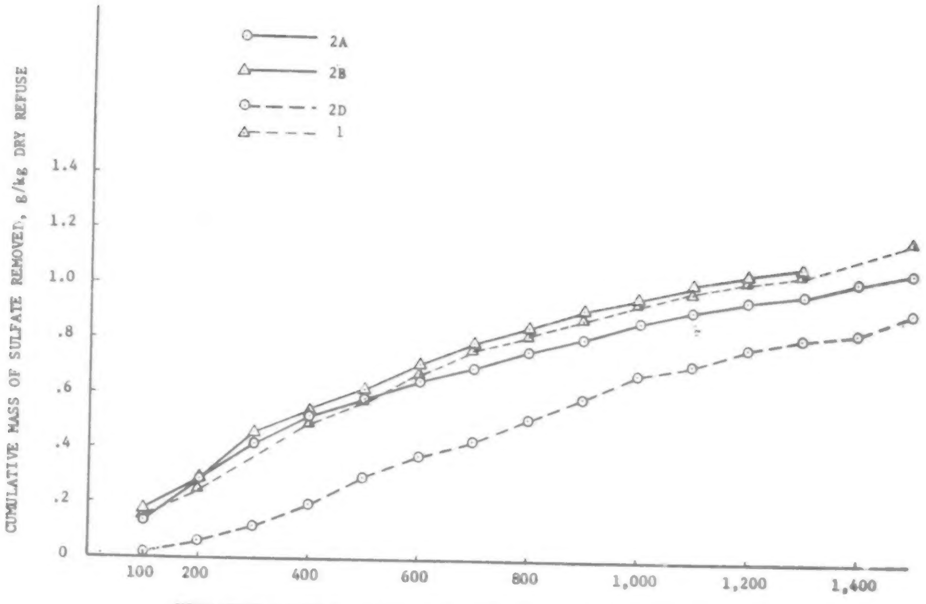

Figure 9 . Cumulative sulfate mass removal.

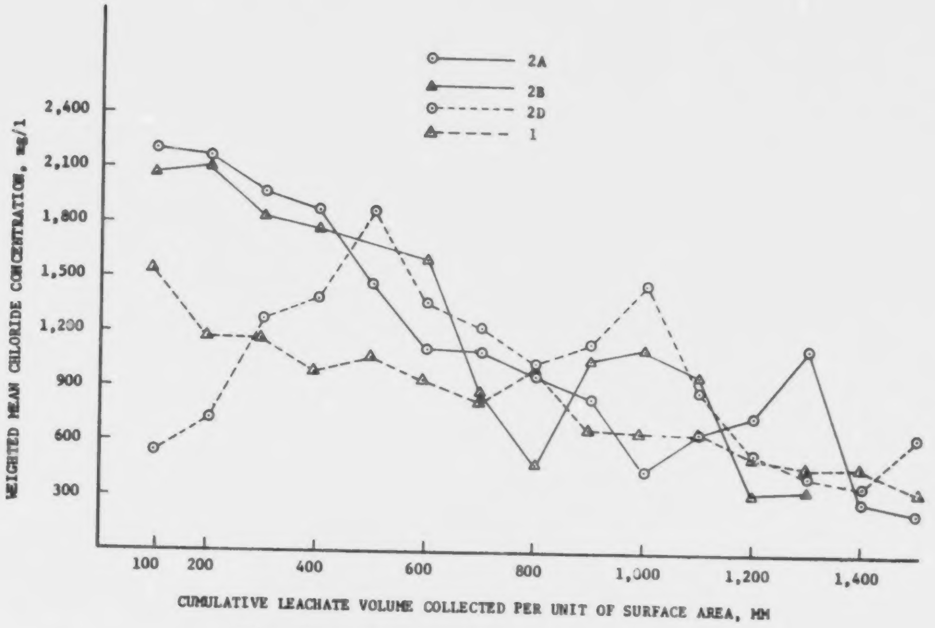

Figure 10 . Weighted mean chloride concentration.

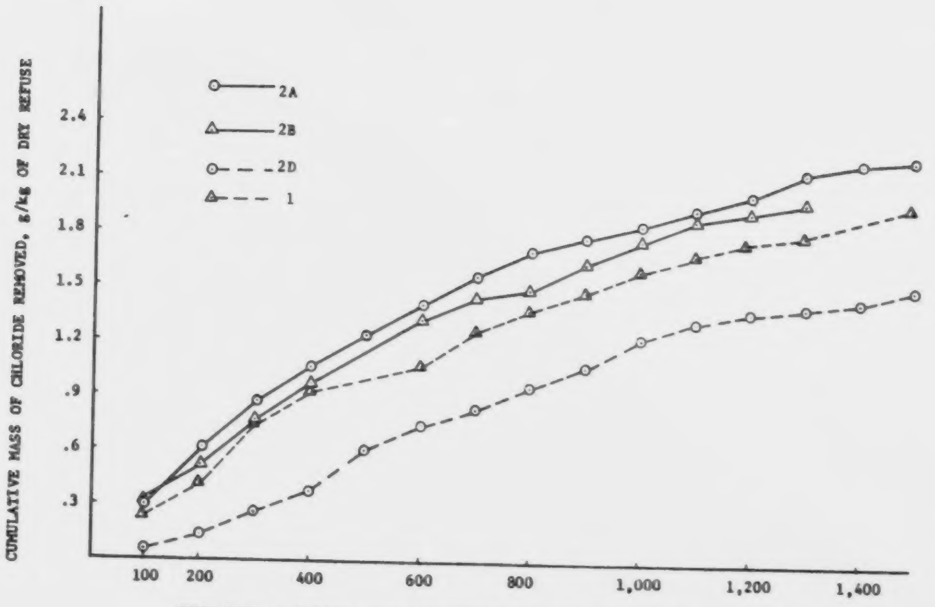

Figure 11. Cumulative chloride mass removal.

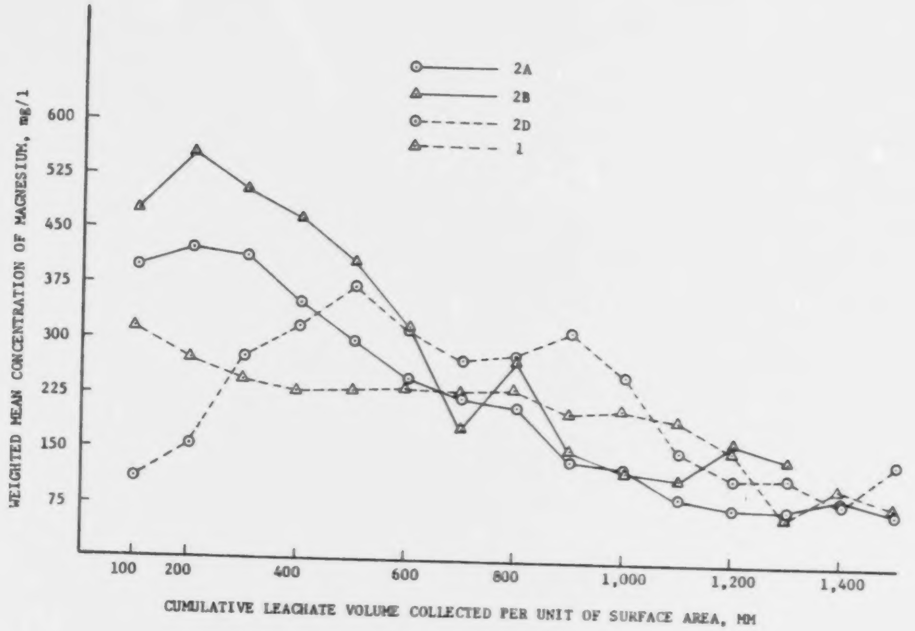

Figure 12. Weighted mean magnesium concentration.

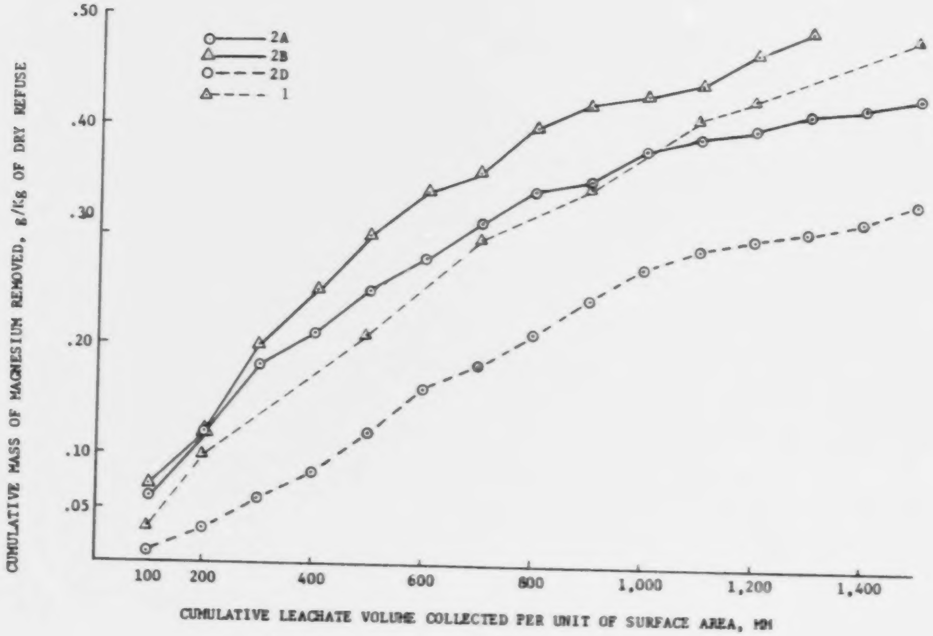

Figure 13. Cumulative magnesium mass removal.

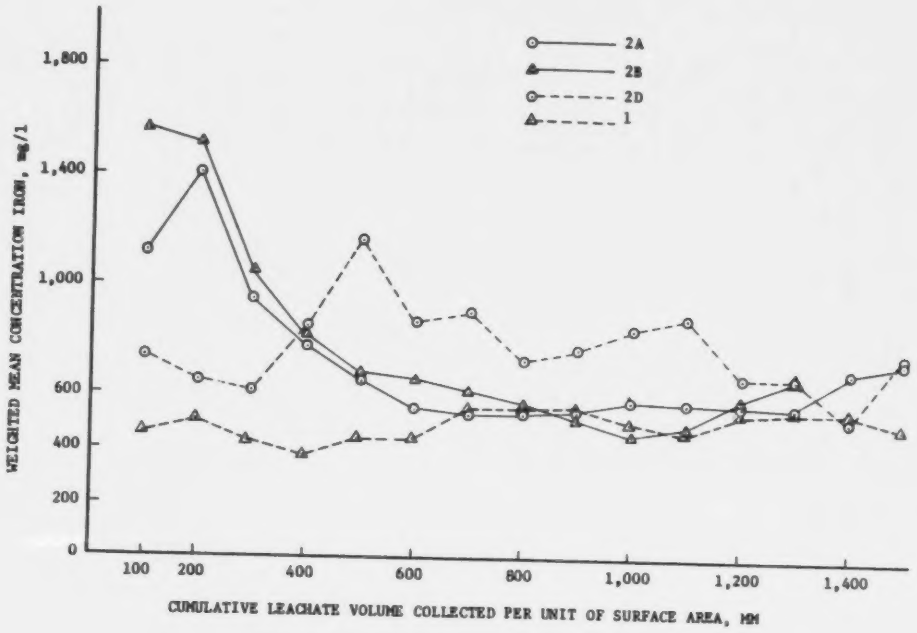

Figure 14 . Weighted mean iron concentration.

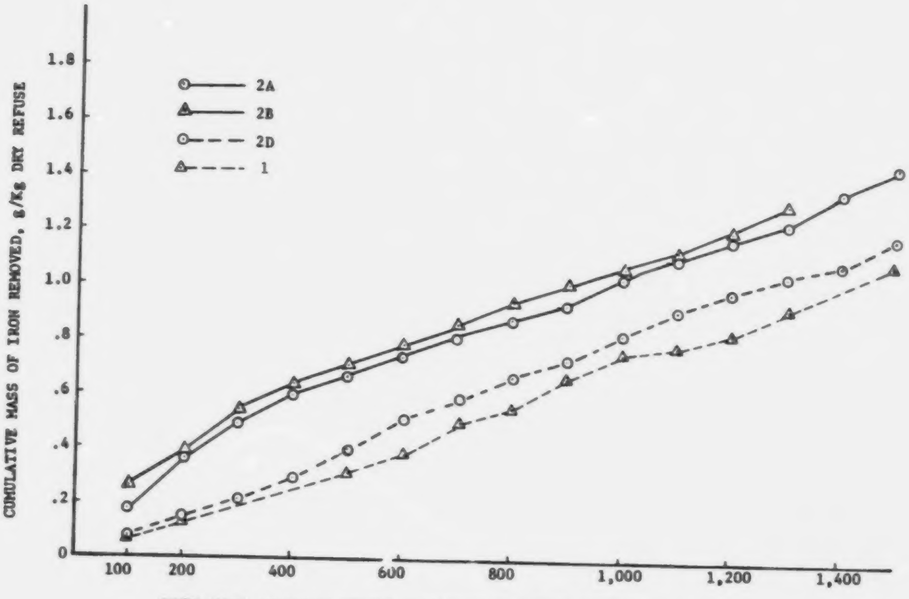

Figure 15. Cumulative iron mass removal.

TABLE 5. PEAK CONCENTRATIONS

Parameter	Test Cell 2A Concentration[a]	Date	Test Cell 2B Concentration	Date	Test Cell 2D Concentration	Date	Test Cell 1 Concentration[a]	Date
COD	57330	7-31-73	61600[b]	4-24-73	41869	11-7-73	37500	1-23-73
Total Kjeldahl-N	1560	7-31-73	1897	10-23-73	1242	10-23-73	364	12-11-72
Ammonia-N	1035	11-7-73	1185	10-23-73	947	11-20-73	536	2-6-73
Ortho-phosphate	390	6-5-73	185	6-19-73	82	7-31-73	61	9-25-72
Sulfate	1306	7-31-73	2000	10-23-73	1280	11-7-73	1160	1-23-73
Sodium	1900	8-14-73	1700	7-17-73	1375	8-28-73	1040	4-3-73
Potassium	2225	7-17-73	2939	11-7-73	1893	11-7-73	1950	6-26-73
Chloride	2335	7-31-73	2343	9-25-73	2260	10-9-73	1749	5-15-73
Iron	1547	8-14-73	2902	4-24-73	1183	9-25-73	616	10-16-73
Magnesium	486	7-31-73	617	10-23-73	411	11-20-73	374	11-27-72
Manganese	109	6-5-73	115	5-8-73	58	8-14-73	184	9-18-72
Calcium	2280	7-17-73	4000	5-8-73	2300	1-29-74	2363	10-16-73
Zinc	150	7-17-73	360	7-17-73	67	7-3-73	104	12-11-72
Hardness	7067	6-19-73	10575	4-24-73	6713	12-16-75	7500	1-23-73
Total Solids	46484	7-31-73	45628	7-31-73	36252	8-14-73	23600	1-23-73
pH	6.2	7-31-73	6.0	12-4-73	6.2	10-2-72	6.3	8-28-71
Alkalinity	11535	7-31-73	13880	2-27-73	8963	2-26-74	8870	3-6-73
Acidity	6720	6-19-73	6843	7-17-73	5057	2-26-74	3620	2-6-73
Conductivity	17000	8-14-73	18000	8-14-73	16000	8-13-74	12200	11-27-72

a. mg/l

b. early peak, concentration later dropped and peaked again on 11-7-73

It was determined that on the basis of a 10 component analysis of composition that variations in performance between cells 2A, 2B and 2C could not readily be attributed to variation in refuse composition. Unit densities of 2A, 2B and 2C were similar, ranging from 392-431 kg/m^3. The refuse depth was the same for all small cells. Water input to each cell was assumed to be the same but the leachate production varied somewhat between 2A and 2B and very widely for 2C.

The initial refuse composition for cell 2D was assumed similar to that of the small cells. The density was somewhat greater at 598 kg/m^3 at a slightly higher moisture content. The possible effect on leachate concentration of this greater density is not known. The input of water was assumed to be the same as to the small scale cells but it was obvious from Figure 5 that it was not. Cumulative leachate production was actually greater than the precipitation. Refuse depth was slightly less in 2D at 2.44 m whereas in the small cells the depth was 2.56 m.

To determine whether the concentration histories were similar for 2A and 2B an analysis of differences was performed. The differences were compared at the 10% significance level.

A summary of the statistical test results comparing the concentration histories of the parameters is presented in Table 6. For only 4 of 14 parameters was the average difference of the concentration history curves of 2A and 2B not significant at the 10% level through the 0 - 1,300 mm range. While 71 percent of the parameters tested indicated test cells 2A and 2B were statistically different (based on the test used) examination of the COD concentration history (Figure 6), as an example, showed very similar responses for the two test cells. This type of data pattern resulted in a small standard deviation, which when divided into the average algebraic difference of the data points, yielded a high significant difference.

A total of 9 of the 14 selected parameters showed no significant difference for the average of 2A and 2B or 2A as compared to 2D. Of the remaining 5 parameters only COD showed a highly significant difference. This indicated that cell 2A statistically provided a reasonable model of 2D for many parameter concentration histories. This is not the situation for 2B where only 6 of the 14 parameters showed no significant difference. Since most of the parameter histories for 2A and 2B were not statistically similar, based on the results of this test it would be inaccurate to state that any model of a large-scale cell would provide as close a resemblance as 2A has to 2D.

Many of the weighted mean concentration history curves showed a later peak concentration for cell 2D than for 2A or 2B with reasonably close or slightly higher concentrations. Since this early large variation in concentrations and time to peak might have influenced the results of the paired difference tests, additional difference analyses were done for the data points on the concentration history curves for 2A and 2D beginning at the peak of the concentration of 2D rather than at the 100 mm data point. These results are presented in Table 7. Only 3 of the 13 parameters did not show a significant difference. It is interesting to note that all of these three, COD, chloride and zinc, were significantly different over the entire data range.

Paired difference testing was also done on the cumulative mass removal curves through 1,300 mm. The results for selected parameters with closely coinciding curves are presented in Table 8. All showed a significant difference, even when two curves almost coincided throughout the entire leachate volume range. While two curves might almost coincide, for example COD, normally one (2A or 2B) was just above the other. This resulted in a small average difference, but a very small standard deviation, and a significant difference.

To overcome this weakness of the test, the gain in mass removal for each 100 mm interval for 2A was compared to that gain for 2B. These paired difference test results are presented in Table 9. This provided a different perspective into the closeness of the cumulative mass removal curves for 2A and 2B with 9 of the 14 parameters showing no significant difference.

These results contrasted with the concentration history results where only 4 of the 14 parameters showed no significant difference. Apparently 2A and 2B were relatively similar in mass removal but not

TABLE 6 WEIGHTED MEAN CONCENTRATION HISTORIES - PAIRED DIFFERENCE TEST RESULTS

Parameter	2A-2B t_{calc}[a]	2A-2B t_{10}[b]	$\frac{2A+2B}{2}$ - D t_{calc}	$\frac{2A+2B}{2}$ - D t_{10}	2A-2D t_{calc}	2A-2D t_{10}	2B-2D t_{calc}	2B-2D t_{10}
COD	3.82	1.83	-	-	5.00	1.83	7.70	1.83
Sulfate	2.17	1.83	-	-	$\underline{0.98}$	$\underline{1.83}$	$\underline{0.47}$	$\underline{1.83}$
K-Nitrogen	4.15	1.83	-	-	1.97	1.83	3.92	1.83
Ammonia-N	$\underline{1.64}$	$\underline{1.83}$[c]	$\underline{1.31}$	$\underline{1.83}$	-	-	-	-
Orthophosphate	2.51	1.83	-	-	$\underline{.921}$	$\underline{1.83}$	$\underline{1.24}$	$\underline{1.83}$
Chloride	$\underline{.943}$	$\underline{1.86}$	2.87	1.86	-	-	-	-
Potassium	$\underline{1.25}$	$\underline{1.83}$	2.14	1.83	-	-	-	-
Sodium	8.87	1.83	-	-	$\underline{1.59}$	$\underline{1.83}$	$\underline{1.70}$	$\underline{1.83}$
Calcium	5.33	1.83	-	-	$\underline{1.69}$	$\underline{1.83}$	5.55	1.83
Magnesium	10.0	1.83	-	-	$\underline{.321}$	$\underline{1.83}$	3.44	1.83
Iron	2.89	1.83	-	-	$\underline{1.45}$	$\underline{1.83}$	$\underline{.259}$	$\underline{1.83}$
Hardness	8.35	1.83	-	-	$\underline{0.79}$	$\underline{1.83}$	5.52	1.83
Manganese	$\underline{.354}$	$\underline{1.83}$	$\underline{1.40}$	$\underline{1.83}$	-	-	-	-
Zinc	7.50	1.83	-	-	2.26	1.83	5.80	1.83

a. t_{calc} - calculated by division of the average difference by the standard deviation of the average differences.

b. t_{10} - tabular t

c. Underscored values shows no statistically significant difference.

TABLE 7. PAIRED DIFFERENCE TEST RESULTS, STARTING POINT AT TEST CELL 2D PEAK OF WEIGHTED MEAN CONCENTRATION HISTORY

Parameter	Starting Leachate Volume[a]	2A-2D t_{calc}[b]	2A-2D t_{10}[c]	Result
COD	500	1.39	1.94	Not Significant
Sulfate	500	10.2	1.94	Significant
K-Nitrogen	600	9.08	2.02	Significant
Ammonia-N	600	10.4	2.02	Significant
Orthophosphate	300	3.79	1.86	Significant
Chloride	500	1.55	1.94	Not Significant
Potassium	500	5.29	1.94	Significant
Sodium	500	5.87	1.94	Significant
Calcium	500	2.10	1.94	Significant
Magnesium	500	4.71	1.94	Significant
Iron	500	9.21	1.94	Significant
Hardness	300	4.50	1.86	Significant
Zinc	300	.983	1.86	Not Significant

a. millimeters - volume per unit of surface area

b. t_{calc} - calculated by division of the average difference by the standard deviation of the average difference

c. t_{10} - tabular t

TABLE 8 . CUMULATIVE WEIGHT HISTORY PAIRED DIFFERENCE TEST RESULTS

Parameter	2A-2B $t_{calc}{}^{b}$	2A-2B $t_{10}{}^{b}$	Results
COD	4.25	1.83	Significant
Sulfate	4.58	1.83	Significant
K-Nitrogen	1.96	1.83	Significant
Ammonia-N	14.2	1.83	Significant
Orthophosphate	63.6	1.83	Significant
Chloride	9.16	1.86	Significant
Potassium	3.14	1.83	Significant
Sodium	29.0	1.83	Significant
Calcium	27.3	1.83	Significant
Magnesium	17.9	1.83	Significant
Iron	8.82	1.83	Significant
Hardness	18.1	1.83	Significant
Manganese	7.21	1.83	Significant
Zinc	44.5	1.83	Significant

a. t_{calc} - calculated by division of the average difference by the standard deviation of the average difference

b. t_{10} - tabular t

TABLE 9 . INTERVAL MASS REMOVAL PAIRED DIFFERENCE TEST RESULTS

Parameter	2A-2B $t_{calc}{}^{a}$	2A-2B $t_{10}{}^{b}$	Results
COD	.522	1.833	Not Significant
Sulfate	.928	1.833	Not Significant
K-Nitrogen	.536	1.833	Not Significant
Ammonia-N	1.35	1.833	Not Significant
Orthophosphate	2.48	1.833	Significant
Chloride	.845	1.860	Not Significant
Potassium	.308	1.833	Not Significant
Sodium	2.78	1.833	Significant
Calcium	1.49	1.833	Not Significant
Magnesium	1.93	1.833	Significant
Iron	.494	1.833	Not Significant
Hardness	2.55	1.833	Significant
Manganese	.405	1.833	Not Significant
Zinc	10.0	1.833	Significant

a. t_{calc} - calculated by division of the average difference by the standard deviation of the average difference

b. t_{10} - tabular t

so in concentration histories. This indicated that the removal of these substances from the refuse in 2A and 2B was limited by the amount of the substance readily available for leaching at that time period in the decomposition process. That is the leachate production varied within some range, and the concentration varied from cell to cell, but the mass removal during each interval was comparable.

Application of the paired difference statistic to the leachate data led to the following observations:

1. Based on incremental increases in masses of parameter removed, test cells 2A and 2B, were not significantly different except for orthophosphate, sodium, magnesium, hardness and zinc.

2. The statistic must be used carefully when evaluating the comparability of concentration or cumulative mass histories. While the majority of parameters showed significant differences between cells (except for 2A and 2D), very similar data which were consistently and slightly different were determined to be significantly different because their standard deviation was small. Thus, cells 2A and 2B, designed to evaluate performance duplication of identically constructed and sized test cells were statistically determined to have performed differently. The test cells, however did perform in a very similar manner.

MATHEMATICAL DESCRIPTION OF LEACHATE CONCENTRATION HISTORY

The repetitive shape of the leachate concentration curves and similar volumes at peak concentrations for many of the parameters indicated that the weighted mean concentration history curves might be mathematically described.

The concentrations in the leachate are dependent on many factors, not all of which are understood or known, especially the magnitude of influence the factors exert on resultant concentrations. Important variables might be the initial mass of a substance readily available for leaching, decomposition of refuse within the fill and subsequent additional mass availability, the pH, solubility limits, the rate of water throughput, decomposition or reduction of the leachate constituents during travel through the fill, the depth of the fill and cover soil effects within the fill. The number of variables involved and the lack of understanding of their effect on resultant leachate concentrations made a semi-empirical curve-fitting approach attractive as an initial effort.

The shape of the weighted mean concentration history curves compared favorably with the equation for two consecutive first order reactions having the form:

$$C = \frac{k_1 k_2 M}{k_2 - k_1} \left(e^{-k_1 v} - e^{-k_2 v}\right) \quad (1)$$

C is the concentration at any volume of leachate, k_1 and k_2 are rate constants and M is the total leachable mass per unit of surface area. The equation is in terms of v, the volume of leachate collected per unit of surface area, to coincide with the weighted mean data and the plotting of concentration versus volume. Since volume of leachate collected is some function of time such as sinusoidal in a seasonal environment, the equation could be changed to a time dependent function.

Since M was unknown, to solve for the rate constants it was necessary to divide Equation 1 by itself when the values of C and v were known, resulting in the form:

$$C = C_{max} \frac{e^{-k_1 v} - e^{-k_2 v}}{e^{-k_1 v_{max}} - e^{-k_2 v_{max}}} \quad (2)$$

C_{max} was selected to be the peak concentration on the weighted mean concentration history curves occurring at cumulative volume v_{max}. k_1 and k_2 were deter- by trial and error for best visual fit to the 2A plot. The respective C_{max}, v_{max}, k_1 and k_2 for selected parameters are presented in Table 10. The comparative concentration history plots are shown in Figures 16-20.

A reasonably good visual fit was obtained with Equation 2 for four of the parameters. It was not possible with this equation to describe the concentration behavior of iron because of the rapid fall after peak. This also happened to a lesser extent with COD and sulfate. Curve fitting in these instances was primarily done so that good coincidence was obtained at the end of the data.

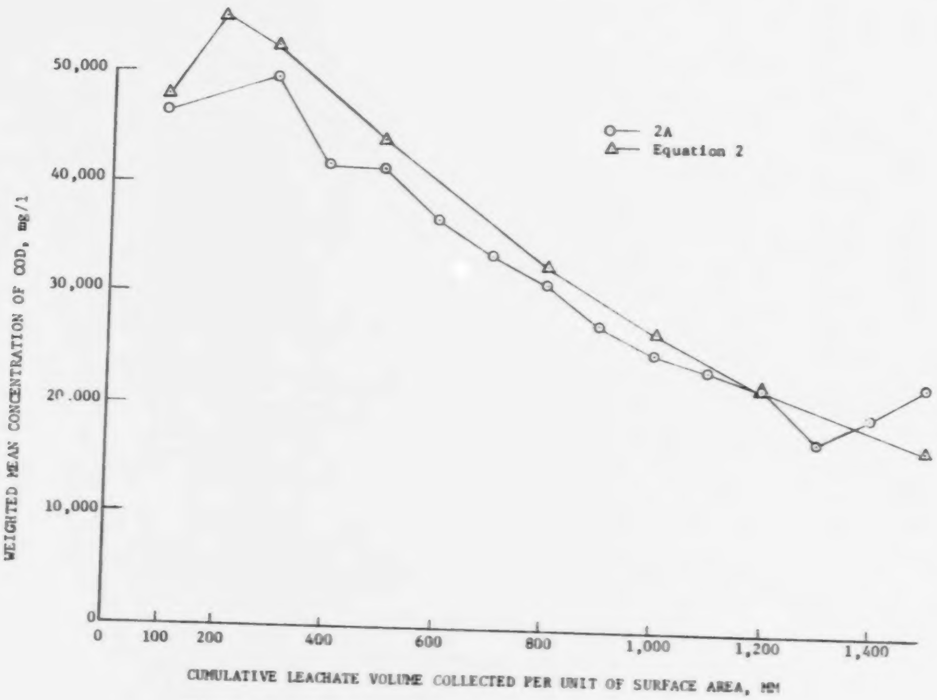

Figure 16. Weighted mean COD concentration history: Eqn. 2 - 2A.

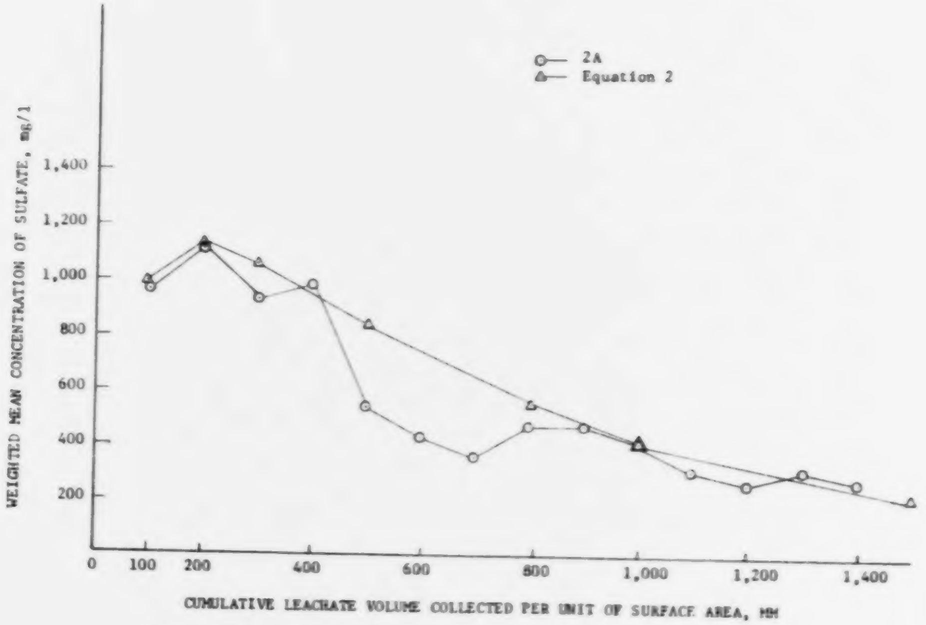

Figure 17. Weighted mean sulfate concentration history: Eqn. 2 - 2A.

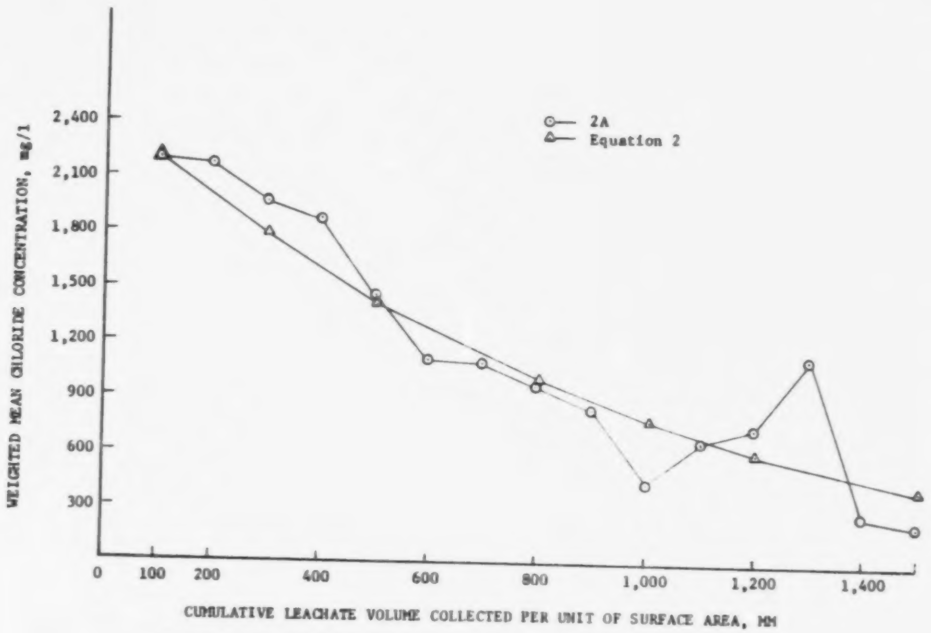

Figure 18. Weighted mean chloride concentration history: Eqn. 2 - 2A.

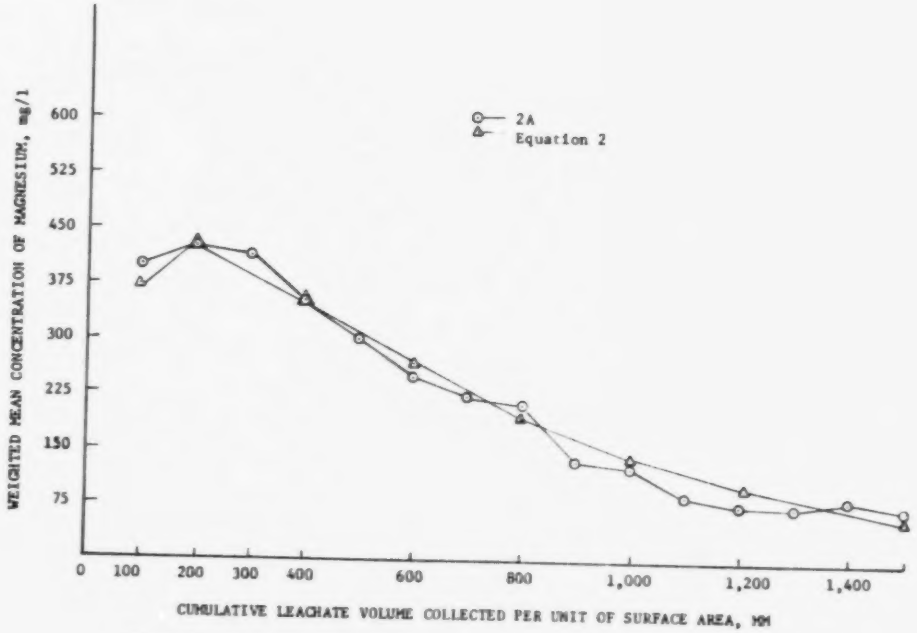

Figure 19. Weighted mean magnesium concentration history: Eqn. 2 - 2A

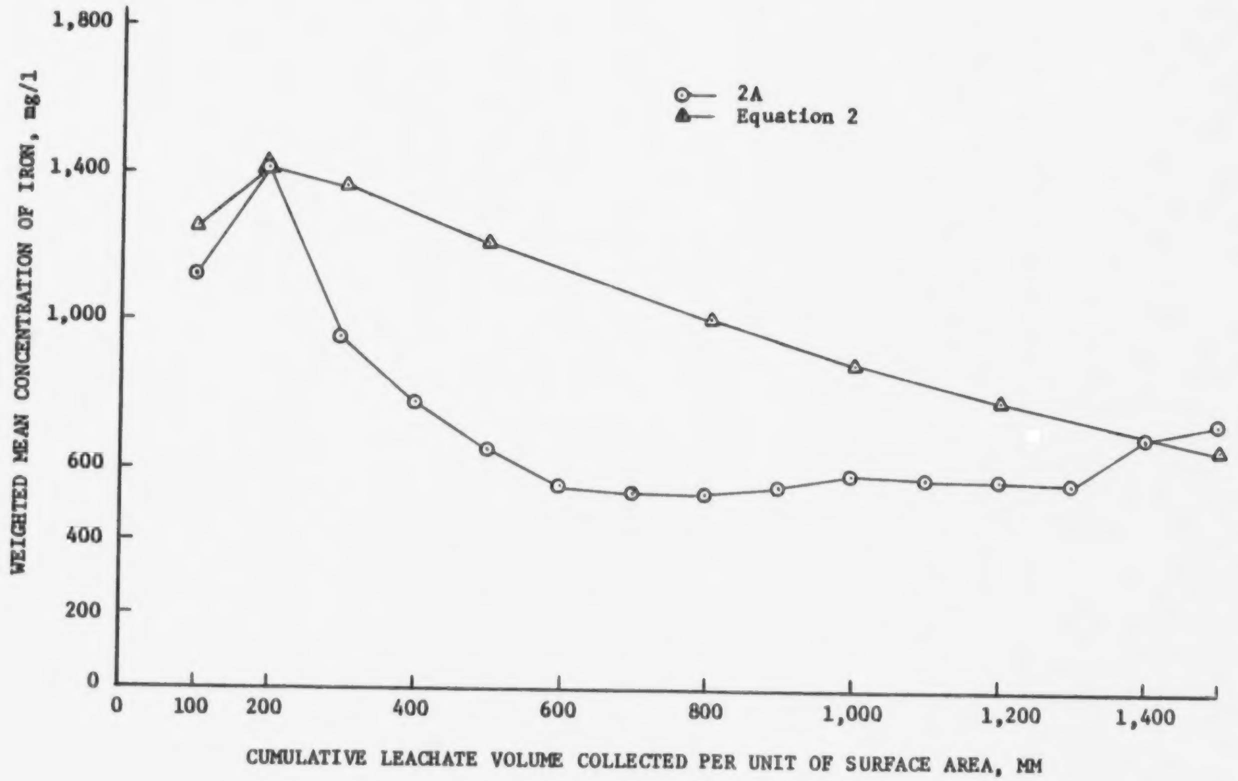

Figure 20. Weighted mean iron concentration history: Eqn. 2 - 2A.

TABLE 10. EQUATION CONSTANTS FOR CELL 2A

Parameter	C^{a}_{max}	V^{b}_{max}	k^{c}_{1}	k^{c}_{2}
COD	55,400	200	.00098	.0145
Sulfate	1,130	200	.00138	.0125
Chloride	2,205	100	.00120	.0350
Magnesium	425	200	.00150	.0120
Iron	1,409	200	.00062	.0170

a. mg/l
b. mm - volume per unit of surface area
c. 1/mm

With k_1 and k_2 known, it was possible to solve Equation 1 for the total mass of a parameter that might be leached. The total mass per kilogram of dry refuse for test cell 2A obtained from Equation 1 is listed in the first column of Table 11. The percent of this calculated total mass that had actually been removed at 1500 mm of leachate is presented in the second column.

The total mass available and percentage removal in Table 11 was based on the shape of only 1500 mm of an infinite concentration history so these values should be treated with caution. It is interesting to note, though, that if this mass is an accurate representation, the very high removal percentages that had occurred after only 1500 mm of leachate.

The total masses projected in Table 11 for sulfate, chloride and magnesium ranged from 1/3 to 1/2 of the total amounts actually measured in samples of the refuse in test cell 1. The projected amount of COD is only 11% of that measured from the samples of refuse in test cell 1. The values in Table 11 might be more accurate projections of the mass that can be removed than that total mass measured in test cell 1 because some of the contaminant mass is probably permanently bound in a non-water soluble state.

SUMMARY AND CONCLUSIONS

Test Cell 1

The initial sanitary landfill test cell was constructed at the Boone County Field Site in June, 1971. It contained 395 metric tons of refuse at a wet density of 592 kg/cubic meter. The maximum depth of refuse was 2.6 m. Summarized findings, based on the data collected through 52.5 months were as follows:

1. Over 791,000 liters of leachate had been collected by December 1, 1975, representing 1831 mm of infiltration through the refuse. It is estimated that an

TABLE 11. CELL 2A - TOTAL AVAILABLE MASS REMOVALS

Parameter	AVAILABLE TOTAL MASS[a]	REMOVAL at 1500 mm
COD	89.6	70%
Sulfate	1.41	75%
Chloride	2.73	81%
Magnesium	.500	86%
Iron	3.26	31%

a. g/Kg of dry refuse

additional 106,000 liters of water was required to bring the cell to field capacity meaning 36.5% of the precipitation over the surface area of the final refuse lift infiltrated the final soil cover into the refuse. Less than 4,000 liters of leachate, .5% of the total volume, passed through the compacted soil liner.

2. The water balance method estimated leachate production at 87.3% of the actual volume collected. Seasonal differences in the actual volumes and estimates were 4% less during the winter (Dec.-Feb.), 19% more in the spring, and 38% more in the summer and fall.

3. The normal concentration history for most of the parameters was low initial values rising to peak concentration during the period field capacity was reached followed by a slow downward trend to 10-35% of the peak value after 4.5 years.

4. Over 58 kg of COD, 1.3 kg of sulfate, 10.7 kg of hardness, 1.3 kg of iron, 1.5 kg of sodium and 2.1 kg of chloride per metric ton of dry refuse had been leached out of the refuse mass after 1830 mm of leachate.

Test Cells 2A, 2B, 2C and 2D

Four sanitary landfill test cells containing municipal solid waste were constructed at the Boone County Field Site during August 1972. Three of the cells, 2A, 2B and 2C were small-scale and the fourth cell, 2D, was constructed similarly to a normal landfill cell. These units were constructed to compare the performance of small-scale systems with a field-scale cell and to evaluate the variations within the three small cells. Findings after 51 months of data collection were:

1. The initial refuse composition and moisture in all cells was determined to be statistically similar. In-place refuse densities in the small cells varied from 392-431 kg/m^3. The density in cell 2D was 598 kg/m^3.

2. After precipitation input to all cells of 2050 mm, leachate collected per unit of surface area varied from 213-2347 mm. The low value occurred in 2C and was probably due to a leak from the cylinder side or base. The upper value was collected from 2D and was in excess of precipitation, indicating leakage into the cell from the soil walls. The apparent field capacity of the refuse in all cells was found to be within 10% of values reported in the literature.

3. Leachate composition histories were statistically compared at intervals of 100 mm of leachate collected using a paired difference test. The statistical test was not adequate, indicating non-similarity where data trends were close and similarity in some instances where graphs showed obvious differences. For most of the parameters studied the concentration histories of 2A and 2B showed similar responses and trends. The histories from 2D characteristically showed a later and lower peak value than in 2A and 2B.

4. Mass removals from 2A and 2B were generally similar and statistically the same on an incremental basis for 9 of 14 parameters. Mass removed per kg of dry refuse from 2D was less than that from 2A and 2B for all parameters examined. After 1500 mm of leachate, mass removal was estimated for five parameters to be from 31-86% of the total available.

5. The density and leachate volume differences precluded a definitive comparison of the behavior of small and large-scale test cells. The statistical evaluation of the comparative behavior of identically constructed small-scale cells was inconclusive.

ACKNOWLEDGEMENT

The test cell data analysis was performed under contract with the Solid and Hazardous Waste Research Division of the Municipal Environmental Research Laboratory in Cincinnati, Ohio. Project Officers for the two contracts were Richard A.Carnes and Dirk R. Brunner.

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INFLUENCE OF MSW PROCESSING ON GAS AND LEACHATE PRODUCTION

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ABSTRACT

A study was initiated in 1974 to investigate the effects of preprocessing municipal refuse prior to landfilling on the decomposition products of that refuse once buried. The preprocessing procedures considered are shredding, baling, or a combination of both techniques. The leachate volume and composition as well as the composition of the gases produced are being considered. The original project was for a three-year period. The project being reported on is a new contract which calls for continued monitoring and data analysis. The study of the gas and leachate was accomplished through the use of landfill simulators buried in the ground. The facility is located at Franklin, Ohio.

The data to date indicates that preprocessing of waste does indeed have effects on the production and composition of leachate and gas. Whether the effects produced are a help or a hinderance will depend on the situations as faced by the individual landfill operator or designer.

INTRODUCTION

The landfilling of municipal refuse is the primary disposal methodology in general use to date. However, the interment of wastes with its subsequent decomposition produces by-products which offer the distinct potential for untoward effects on the surrounding environment.

Leachate production is one of the primary sources for environmental insult arising from landfilled material decomposition. As a concentrated liquid waste stream, it has potential for contaminating both surface and groundwater.

Decomposition gases evolved from a landfill present a potential hazardous situation in the area of the fill site. This hazard is of a long-term nature and the threat exists long after a fill is completed. This is especially true in the case of methane which in concentrations of 5 to 15 percent with oxygen in the air forms an explosive mixture.

This paper concerns itself with the study of the preprocessing of municipal refuse prior to landfilling and its subsequent effect on the leachate and gases produced. The preprocessing procedures considered were baling, shredding, and a combination of both procedures.

CELL OPERATION

Five landfill simulators are located at the SYSTECH Waste Treatment Center site in Franklin, Ohio. They are buried concrete cells each of which contains municipal refuse which had been subjected to different preprocessing steps.[1] The test cells were charged with refuse as follows.

Cell	Refuse	Weight (ww)
1	Baled, shredded refuse	11,700 kg
2	Baled, whole refuse	11,700 kg
3	Baled, whole refuse (saturated)	11,700 kg
4	Unbaled, shredded refuse	10,800 kg
5	Unbaled, whole refuse	9,700 kg

All of the weights listed above were calculated except the refuse wet weights for Cells 4 and 5. The cells were instrumented and plumbed for temperature monitoring, leachate collection, water addition, and gas sampling.

Water additions were made in accordance with a schedule designed to resemble the rainfall patterns of the midwestern region of the United States.

Leachate collection and gas sampling were accomplished via plumbing lines which connected each of the simulators with a centrally located instrumentation cell. Temperature data was gathered through the use of 24 thermocouples placed in each of the simulators.

Leachate samples and gas samples were collected on a monthly basis and analyzed for those parammeters pertinent to sanitary landfill operation. The cells were also quantitatively drained monthly to perform water balance studies.

MOISTURE MONITORING

Cells 1, 2, 4, and 5 have been operated on a water addition and volumetric leachate collection basis for approximately 3.5 years. The water additions were made according to a schedule resembling the annual rainfall pattern for the midwestern United States. This schedule is shown in Table 1. The particulars of the physical arrangement for the water addition procedures were described in depth in the final report under a prior contract with the USEPA (Ref. 1).

During the time period of the study to date, Cell 3 was operated under "saturated" conditions. This was accomplished by only withdrawing enough leachate to perform the required analyses during the initial saturation period. After achieving saturation, the leachate was drained monthly and an amount of water was added which was equal to the volume of leachate withdrawn. As no transpiration or evaporation takes place within these enclosed cells, the amount of moisture in Cell 3 is assumed to remain constant.

The theoretical field capacity of refuse is its absorptive capacity for moisture. In theory, the amount of water infiltrating a landfill should equal the quantity of leachate exiting the landfill once field capacity is reached. In a practical, real world situation, however, the definition of apparent or practical field capacity as reported by Fenn[2] is more apropos to the problem. Fenn defines field capacity as the maximum moisture content which a soil (or solid waste) can retain in a gravitational field without producing continuous downward percolation. The data in this report is presented in terms of "field capacity" having been attained at the initiation of continuous leachate production.

The total moisture retained within a cell is directly inferred from the volume of water added minus the volume of leachate removed. As the test cells are enclosed, transpiration and runoff do not enter into the calculation. The appparent moisture retained in the cell was calculated as follows:

$$\text{Apparent moisture retained in the cell (ml/kg)} = \frac{1000\ \text{water added}\ (\ell) - \text{Leachate removed}\ (\ell)}{(\text{kg solid waste})}$$

As far as reaching the point of continuous leachate production is concerned, Cell 2 (containing baled, unshredded waste) began producing leachate in the second month of operation. Cell 1 (baled, shredded) began producing in the fifth month, Cell 4 (shredded) began in the fifth month, and Cell 5 (unshredded) began producing in the sixth month. The results indicate that the preprocessing of the wastes prior to interment did have some positive effect on the time frame in which continuous leachate production was begun.

The total moisture retained for the cells containing baled refuse (Cells 1 and 2) continued the trend toward similarity, but the cells containing unbaled refuse abandoned the trend exhibited earlier and showed a new trend toward divergence with Cell 5 (unbaled, whole refuse) having a higher propensity to retain moisture. Both unbaled cells retained significatly greater amounts of moisture overall than did the baled cells. The obvious inversely proportional effects of the moisture retained phenomena on leachate volume is shown in Figure 1.

Similar results were observed when the moisture retained was normalized to the wet weight of the refuse landfilled. The average moisture retained figures in

The average moisture retained figures in Table 2 further illustrate this phenomenon.

The data to date indicate that the processing of refuse prior to landfilling does have a positive effect on the start up of continuous leaching as well as the volume of leachate produced when compared to refuse which is landfilled without baling or shredding. This trend was noticeable during the course of the original study and has become even more pronounced during the first year of this effort.

The moisture retained values are still rising again indicating that theoretical field capacity has not been reached.

LEACHATE DATA

Leachate samples have been collected since January 1975. The leachate was withdrawn from the bottom of each test cell via lines which have their terminus in the lower level of the instrumentation cell. Collection was made from each of the cells on a monthly basis. The volume collected was recorded and analytical procedures to determine chemical and biological parameters were carried out.

The pH and conductivity of the leachates were measured and the individual leachate samples were analyzed for chemical oxygen demand, total organic carbon, alkalinity, total phosphorous, Kjeldahl nitrogen, total solids, and for the following metals: iron, copper, zinc, cadmium, chromium, lead, and nickel. In general, the analyses were completed according to methods described in Chian and DeWalle.[3]

General statements that can be made about the leachate emanating from five test cells are that the greatest amount of leachate to date has emanated from the baled cells, Test Cells 1 and 2, closely followed by the saturated bale cell, Test Cell 3. Much smaller total amounts of leachate volume have emanated from Test Cell 4, the Shredded Cell, and even less from Test Cell 5 containing the unprocessed waste. Leachate volume in itself, however, is not a measure of the potential for environmental insult due to the leachate since, in most cases, it can be stated that the "quality" of the leachate is "better" in the Test Cells 1 and 2 and "worse" in the leachate from Test Cell 4. By this we mean that the concentration of measured pollutants in the leachate, as well as the total accumulative mass of specific measured pollutants in the leachate, tends to be least in Test Cells 1 and 2 and tends to be greatest in Test Cell 4.

Because of the slight, but conceivably important differences in waste composition and in leachate volume emanating from the test cells, it was necessary to devise a means of adjusting the viewpoint with which we evaluated the leachate emanating from the test cells. It was decided that for purposes of analysis and interpretation the most meaningful measure of leachate quality would be a measure of the total mass of specific pollutants emanating from each cell per kilogram of solid waste in the cell. Thus, the pollutant out is calculated by multiplying the concentration of pollutant as measured each month, times the volume of leachate for that month, divided by the total mass of solid waste in the measured test cell.

There is some question about the equivalence of even this measured parameter when attempting to compare test cells. It was of some concern as to whether or not the most meaningful measure would be cumulative pollutant as a function of time. This approach would give an indication of the importance, over some period, of the biological activity within a landfill for the different processings as well as the physical and chemical interactions taking place. Alternatively, we could consider the total cumulative pollutant leached as compared with the leachate volume passing through the test cell. This latter measure would seem to give a better indication of the physical and chemical interactions alone since it will be dependent upon the amount of moisture actually flushing through the cells. Admittedly, this would also affect the biological activity, but we feel that, in general, the comparison on a time basis gives measure of relative biological activity and its effects, while the measurement as a function on the leachate volume out, is a more direct indicator of the early stage physical and chemical activity within the test cell. This is not to infer that either approach is more desirable than the other. There are merely two different ways of looking at essentially the same data.

It is commonly accepted that baling will prevent infiltration of moisture into the mass of solid waste, while shredding will accelerate this infiltration. These two mechanical processes have different aims. The intent of baling is to delay the decomposition of solid waste as long as possible so as to result in a weaker and thus more "treatable" leachate. The aim of shredding is to accelerate the decomposition of the waste and to achieve a stable landfill in a shorter period of time. The actual needs and wishes of any locality would determine which of these approaches was more desirable to them. Table 3 shows a comparison of the test cells leachate on a common volume basis at 5200 liters of leachate out. This leachate volume was chosen since it is the greatest amount of leachate to have emated from Test Cell 5 to date and, thus, all cells could be compared on a common leachate volume basis. Viewing the table, it can be immediately noticed that the biological parameter measures of total alkalinity, total solids, total organic carbon, total Kjeldahl nitrogen, and chemical oxygen demand all exhibit similar distinquishing characteristics in comparing processing methods.

Test Cells 1 and 2, both of which are baled, show nearly identical total cumulative values per kilogram for these parameters and they are quite obviously significantly smaller than the totals for any of the other test cells. Test Cell 3, which is operated in a saturated condition, is considerably greater than either Test Cell 1 or 2, although it also is baled and, as a matter of fact, shows almost identical cumulative leachate parameters as Test Cell 5 for these biological measures. We are confident that this similarity with Test Cell 3 and 5 is happenstance, although it is a fortuitous happenstance, and may serve as a guideline for the nature of pollutants emanating from a baled cell in an exceedingly moist environment. The final conclusion to be drawn on the biological parameters is that the shredded material in Test Cell 4 does indeed seem to be stabilizing at an accelerated rate since the total amount of each measured parameter is significantly greater than the leachate of the unprocessed or the baled cells.

It is interesting to note that not only in the biological parameters but, as may be seen in the rest of Table 3, also in the metal parameters measured, Cell 1 which contained shredded baled waste is in every case nearly identical with Cell 2 which contains only baled but otherwise unprocessed waste. This would seem to indicate that baling of the waste is the important physical process in this case and that the addition of shredding of the waste (unless it were to achieve easier or cheaper processing) is superfluous and unnecssary for baling. In other words, if a community is intending to operate a baled landfill and is contemplating the purchase of a shredder in addition to a baler, this second piece of equipment would appear to be unnecessary if the long-term, slowly stabilizing situation of a baled landfill is desired. If rapid stabilization is desired, then we would recommend not baling the waste but rather shredding alone. The combination of the two technologies shows no measurable advantage to date.

The chemical parameters measured consistently throughout the study were selected metal ions. Table 3 thus also shows, on a common leachate volume basis, the amount of iron, copper, cadmium, zinc, nickel, chromium, and lead leaching from the five test cells. The results for these materials are not as readily interpreted as the biological parameters. It is noticeable that the amount of metals leaching from the shredded cell is in nearly every case much greater than the amounts leaching from the other cells. It is also noticeable that Cells 1 and 2, the baled only and shredded baled cells, are nearly identical in these parameters, but the strong statements which are obvious for the biological parameters are not nearly as obvious, for the metal constituents of leachate.

It is thought that the effect of processing on the biological parameters is much more noticeable because these tend to be measures only of the organic material contained in the waste which is very similar both in composition and quantity. The components in the solid waste, however, which can result in leaching of specific metals, are present only in small amounts and thus are highly susceptible to great variability from one sample to the next. It is our feeling at this time that the inherent variability of solid waste has resulted in large variations of specific metal-bearing wastes from cell to cell. This is because the total leachable metal ions, although small in the refuse, are susceptible to high percentage variation. In other words, if the waste in Cell 1 were to contain 0.01 percent of its total mass as zinc salts and Test Cell 4 were to contain 0.03 percent of its total mass as zinc salts, both of these are exceedingly small numbers, difficult to measure with any great accuracy, and yet note that one is three times as great as the other. Such slight discrepancies can result in tremendous variation in the quality of the leachate in each case. This points up one of the more severe drawbacks to the sort of uncontrolled experiment which is common practice in evaluating and experimenting on landfilling as a waste disposal methodology. Even within this limitation, it is obvious that the shredded cell produces higher levels of each measured pollutant while the baling seems to produce a more dilute leachate in each case measured on a basis of common leachate volume. This, in both cases, is thought to be due to the nature of the processing employed and not to the admitted variations in the composition of the waste in the test cells. The shredding obviously makes more active sites available both for biological activity and for chemical and physical interactions. This is born out by the fact that the shredded cell has retained a much higher level of moisture than the baled cells, and in fact, the waste in Test Cell 4 seems to act in a manner similar to a household sponge.

Figures 2 through 25 show the plots of cumulative masses of leachate parameters versus, in one case, cumulative leachate volume and, in the other case, time.

GAS COMPOSITION DATA

The gases produced via refuse decomposition in a landfill include both the products of aerobic decomposition (CO_2) and anaerobic decomposition (CH_4 and CO_2). The bacteria responsible for methane producton are strict anaerobes. Therefore, methane will not be produced when oxygen is present. The fact that researchers (4, 5), have found oxygen and methane simultaneously in their samples is an indication that both situations can occur simultaneously but in different portions of the landfilled waste. Such factors as oxygen presence in the upper layers of a landfill, dissolved oxygen in the infiltration water, and the heterogenous nature of the refuse results in pockets of anaerobiosis and aerobiosis occurring simultaneously in the same general landfill area.

The process of anaerobic decomposition is similar to that found in an anaerobic sewage sludge digester. The initial process is the formation of organic acids by the facultative anaerobic acid formers. These organic acids then provide a suitable substrate for the anaerobic methane formers which utilize them to produce methane. Other landfill researchers (6) report this process to be occurring in landfills. One of the important goals of this study was to determine compositional data on the gases generated in a landfill environment and the effects of preprocessing of waste on gas composition.

Figures 26, through, 29, show time history of the composition of gases drawn from the test cells throughout this study. The first impression is one of lack of consistency both in time and from cell to cell. Data indicative of high-methane production is the situation for cell one containing the baled, shredded waste. It is obvious from the data that this cell is and has for some time been indicative of an anaerobic methane-producing situation. Data for Cell 5, however, indicated that little or no methane had been produced within this cell to date. This is a difficult conclusion to accept in light of the fact that it would seem reasonable that at least some portion of such a large waste sample should have achieved anaerobiosis. However, this is not borne out by the data. The other cells all show varying levels of anaerobiosis achieved. Variations in the composition of the gas stream with time are not readily explainable either in terms of seasonal changes or other control factors such as water addition rate.

In previous reports we indicated that the early data did seem to follow seasonal variations in the gas composition. At that time we had hypothesized that this could be due both to variations in the temperature of the waste and its concomitant effect on the health of the anaerobic culture as well as the dissolved oxygen in the water added to the cells which also would have a very important effect on the anaerobic cultures. Later data, however, does not seem to bear out either of these hypotheses and, thus, we have no ready explanation for the time variations in the gas composition which are seen.

CONCLUSIONS AND RECOMMENDATIONS

Analysis of the data collected in the study during the time period covered by this report gives rise to the following conclusions.

In regard to moisture infiltration of a landfill, the data indicate that the preprocessing step of baling shows an enhancement of leachate production both in regard to time passage to the initiation of continuous leaching and in the overall volume of leachate produced. This is shown by the fact that the baled cells continuously produced leachate three to four months before the unbaled cells. It also can be seen in the total leachate collected over the test period as well as the moisture retained in the refuse cells during the test period. It is concluded that the baling step, not the shredding, is the processing procedure to consider where early or heavy leaching is of concern.

The pollutant concentration, when compared cell to cell on an equal volume leached basis, shows that shredding is the processing procedure that has the greatest enhancement effect on leachate pollutants produced. Baling on the other hand substantially lowers the pollutant mass flow from a landfill.

Taken together, these observations lead to the conclusion that the baling of refuse will allow for a landfill which will stabilize over a long period of time with a greater volume of less concentrated leachate being produced during the process.

On the other hand, shredding appears to offer the shorter stabilization time frame but with a lesser volume of more concentrated leachate being produced.

The data concerning the baled waste in a saturated environment when compared with unprocessed waste offer an coincidental but interesting conclusion. That is, the baled waste in the saturation state behaves quite similarly to unprocessed waste in a normal landfill environment in regard to pollutant concentration produced. This finding should be of special interest to designers of landfills in geographically wet areas.

In summary, based on the data to date, it appears that either baling alone or shredding alone are the preprocessing procedures which should be considered when designing landfills. Combined baling and shredding appears to offer no advantages when considering the quantity or quality of leachate produced.

In regard to the composition of gases produced in landfills with preprocessing, the only conclusion to be drawn at this time which is supported by the data is that the shredding of waste will result in a much more rapid methane production.

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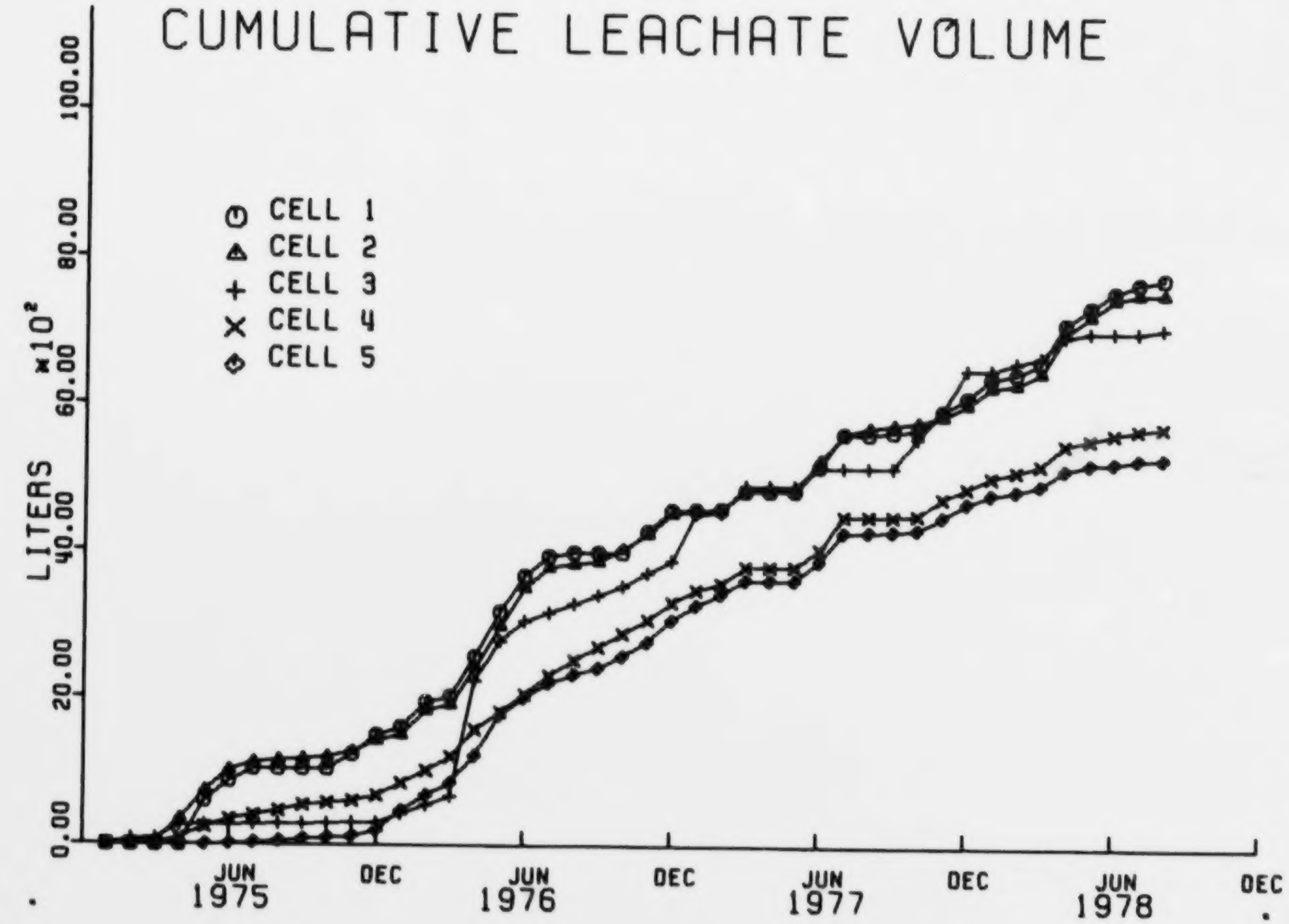
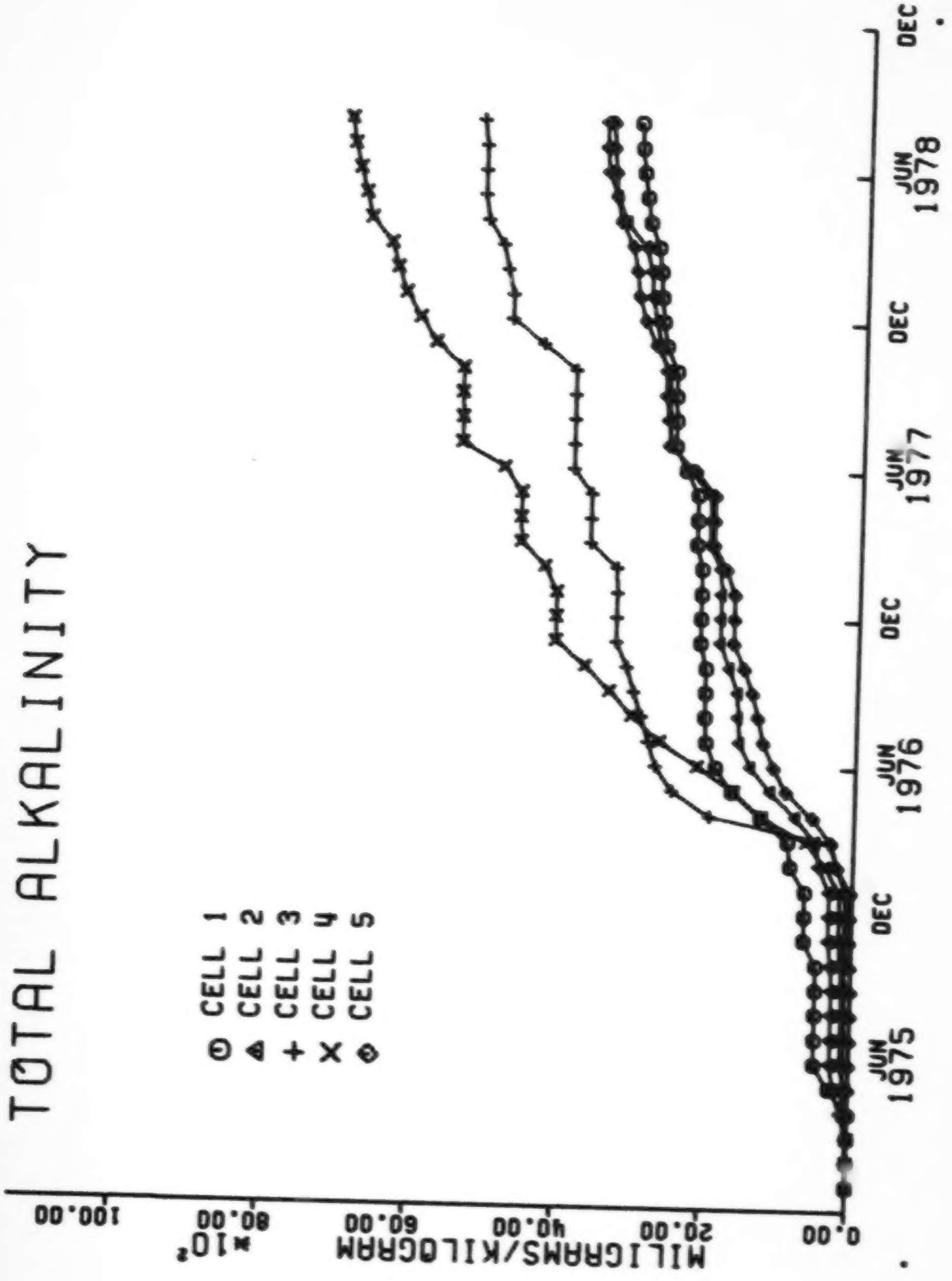
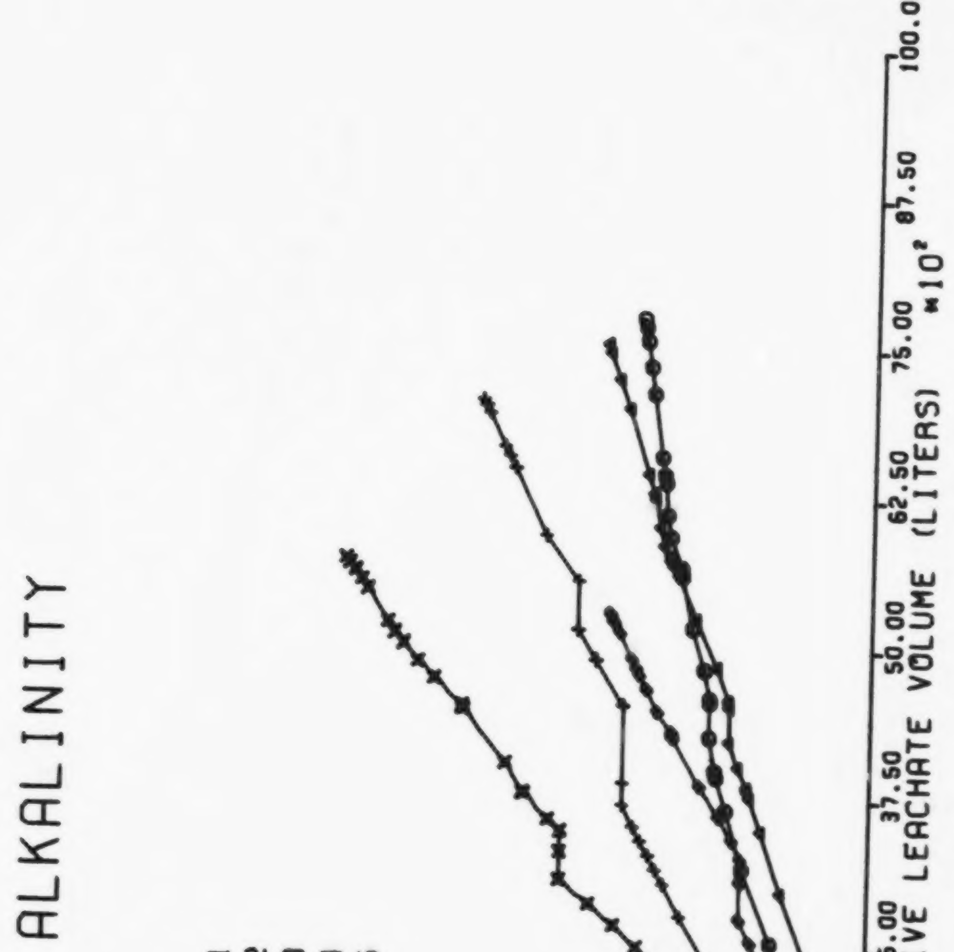
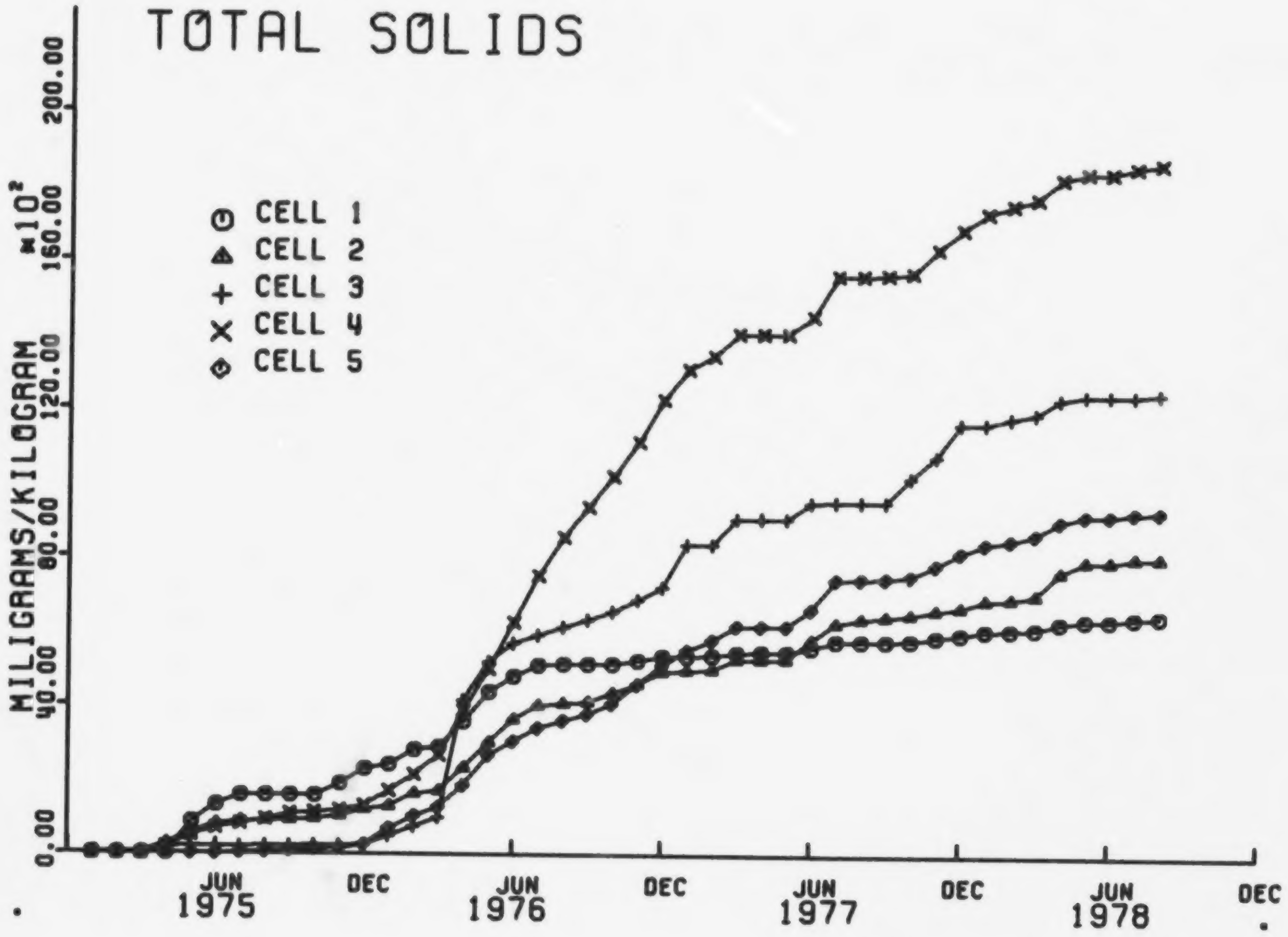
Table 2. Water Balance Summary
Table 3. Comparison of Mass Flow of Leachate Parameters On An Equal Leachate Volume Basis.

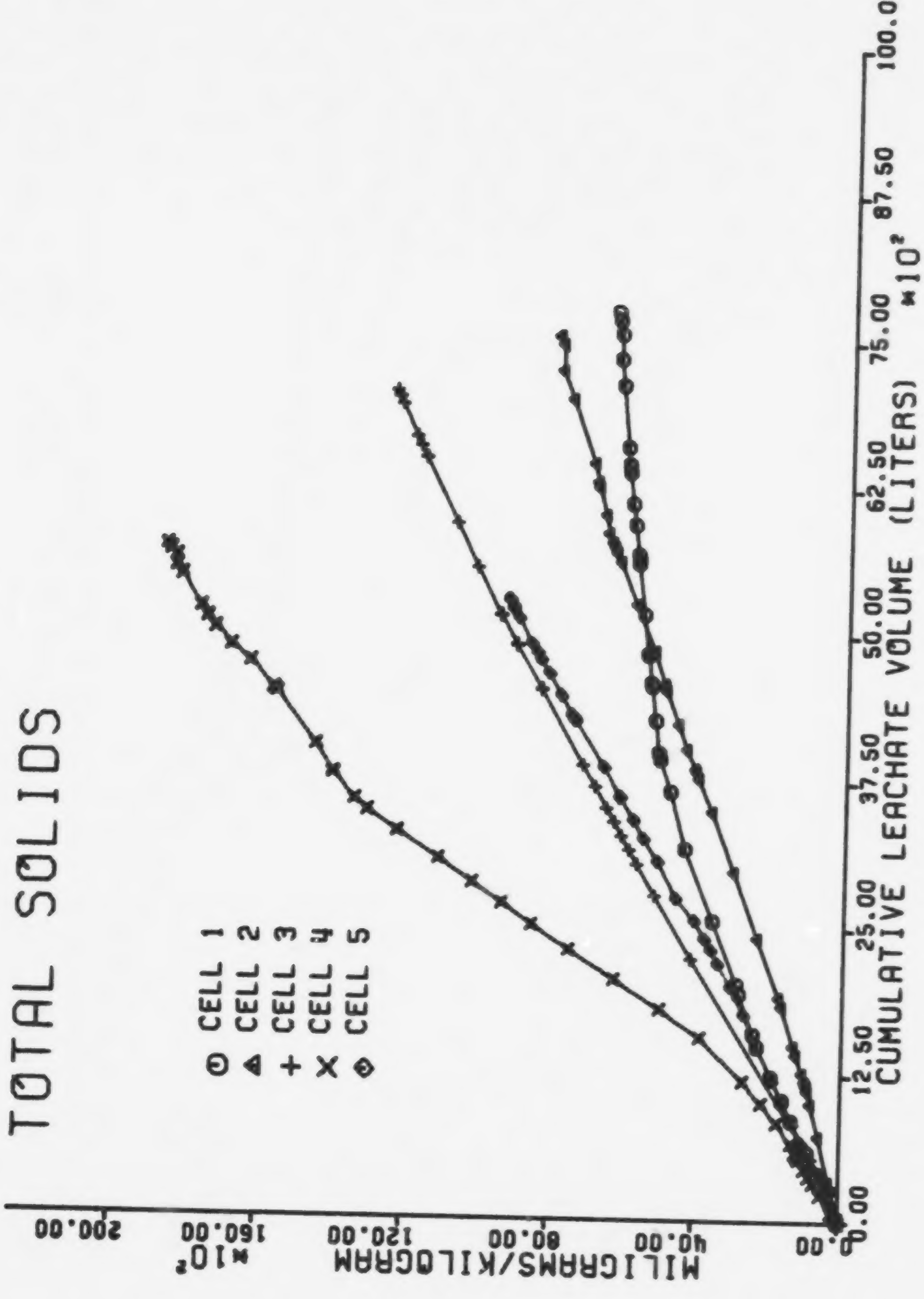


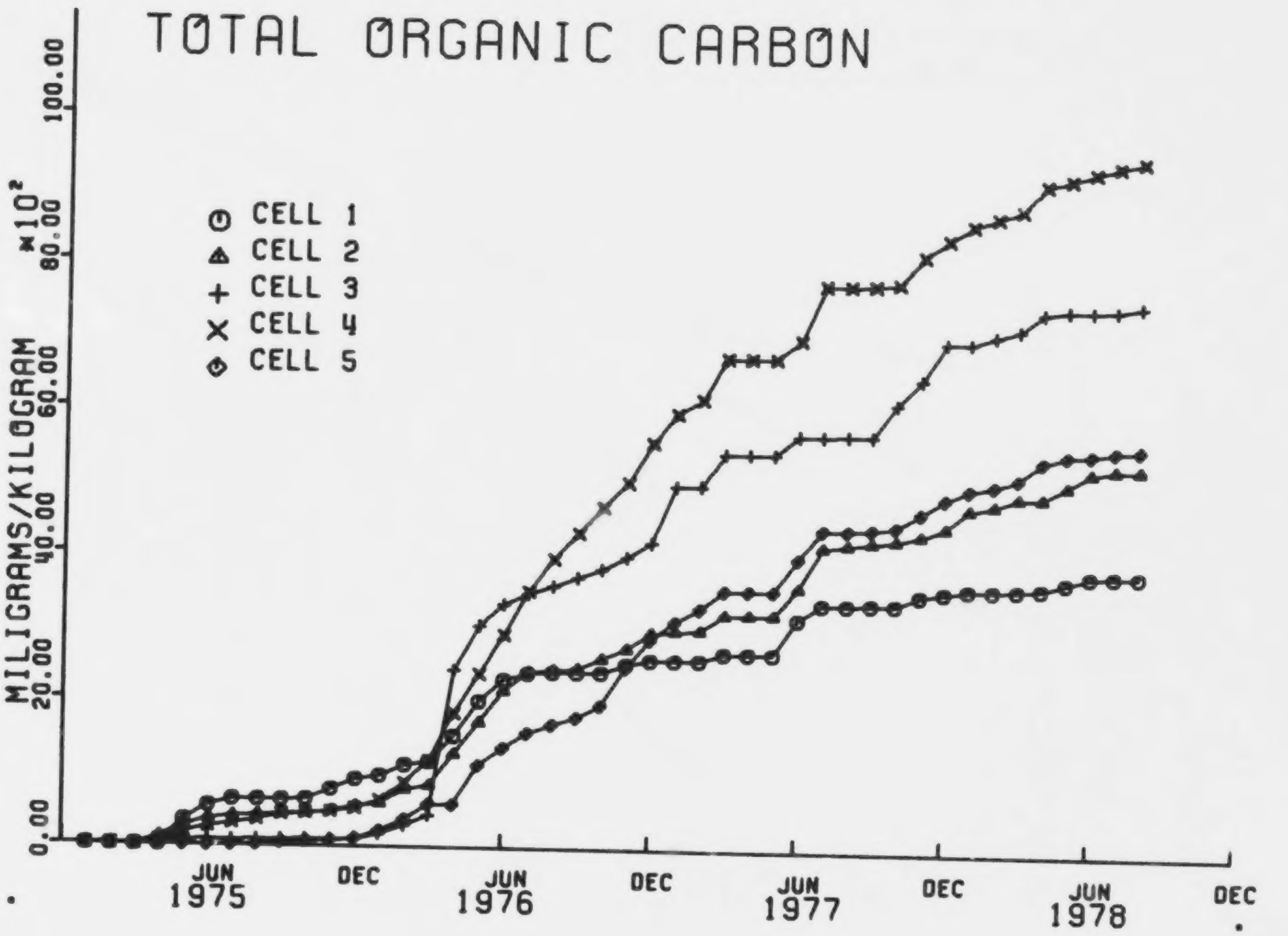
CUMULATIVE LEACHATE VOLUME

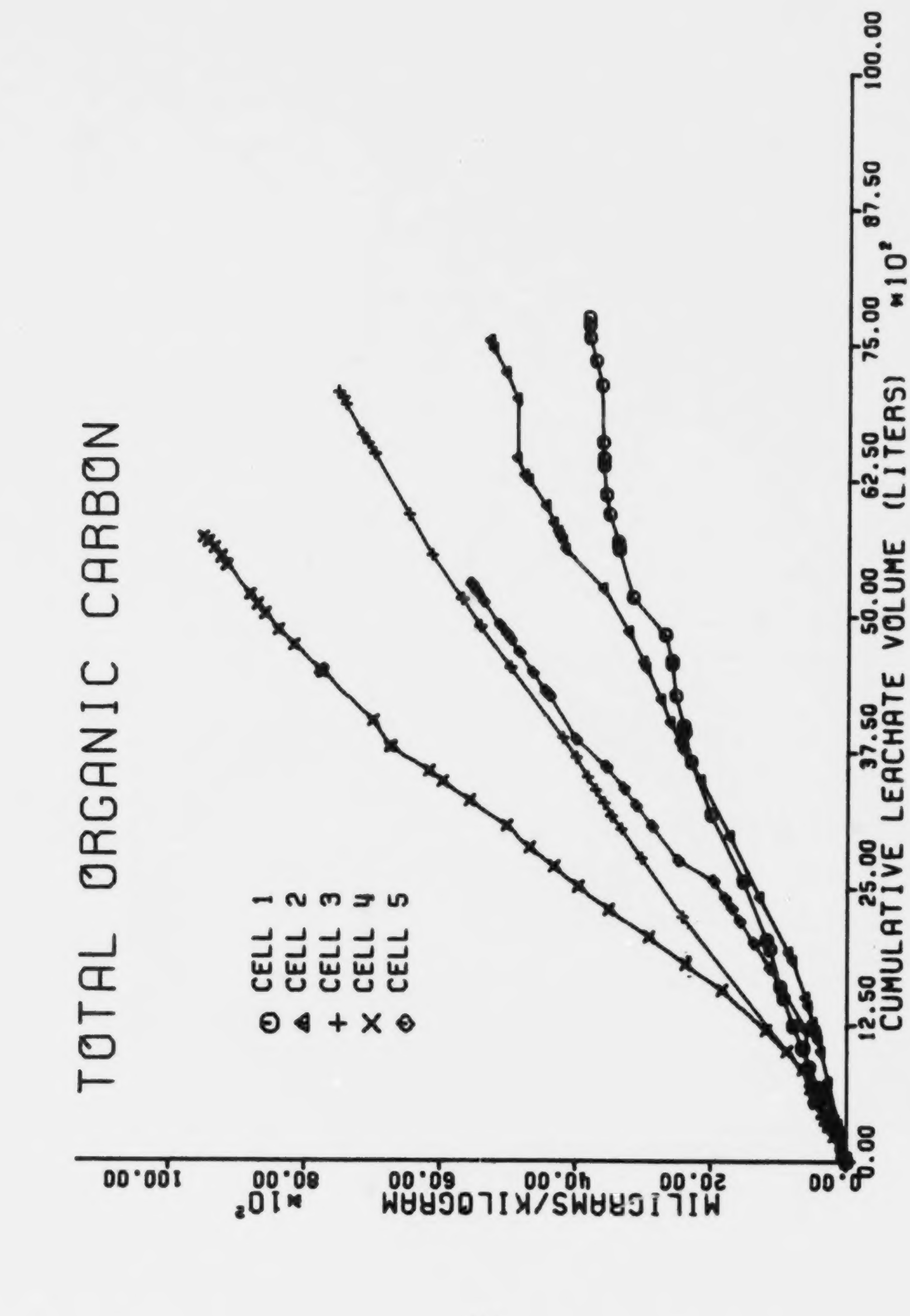

TOTAL ALKALINITY
⊙ CELL 1
▲ CELL 2
+ CELL 3
X CELL 4
◈ CELL 5
MILIGRAMS/KILOGRAM
*10²
0.00
20.00
40.00
60.00
80.00
100.00
JUN 1975
DEC
JUN 1976
DEC
JUN 1977
DEC
JUN 1978
DEC

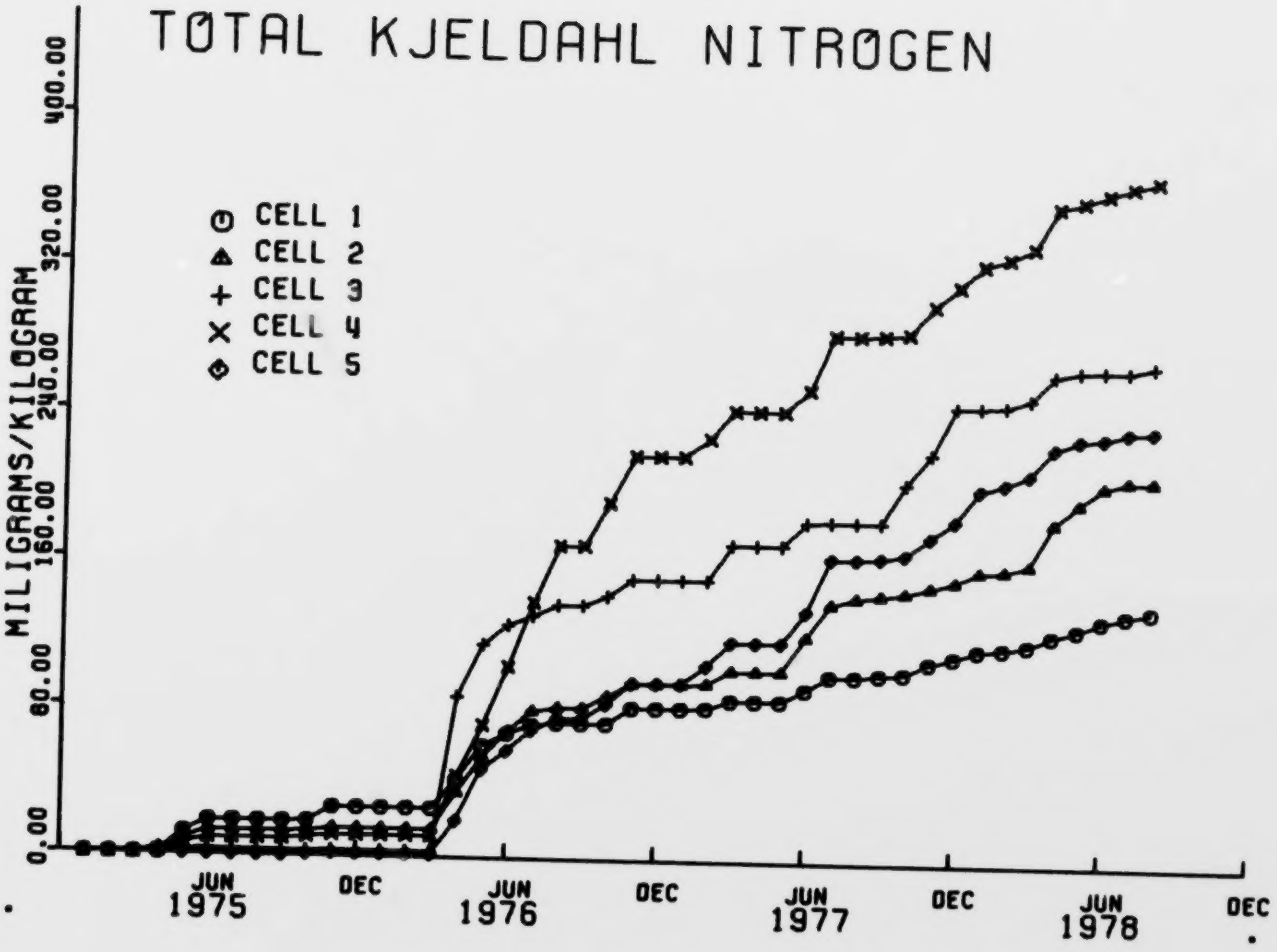

TOTAL ALKALINITY
⊙ CELL 1
▲ CELL 2
+ CELL 3
X CELL 4
◇ CELL 5
MILIGRAMS/KILOGRAM
×10²
0.00
20.00
40.00
60.00
80.00
100.00
CUMULATIVE LEACHATE VOLUME (LITERS) ×10²
0.00
12.50
25.00
37.50
50.00
62.50
75.00
87.50
100.00

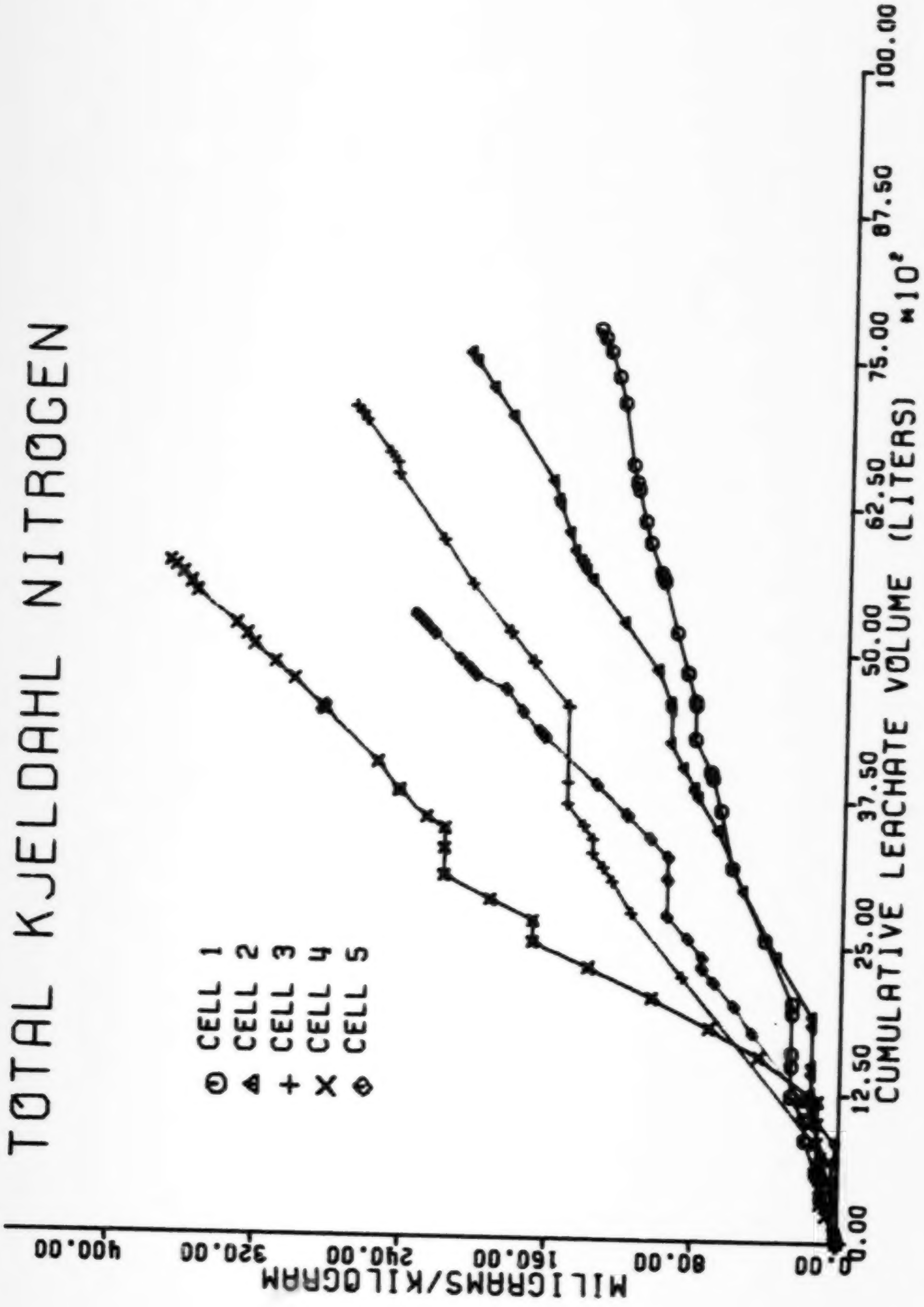

TOTAL SOLIDS
CELL 1
CELL 2
CELL 3
CELL 4
CELL 5
MILIGRAMS/KILOGRAM
$\times 10^2$
0.00
40.00
80.00
120.00
160.00
200.00
JUN 1975
DEC
JUN 1976
DEC
JUN 1977
DEC
JUN 1978
DEC

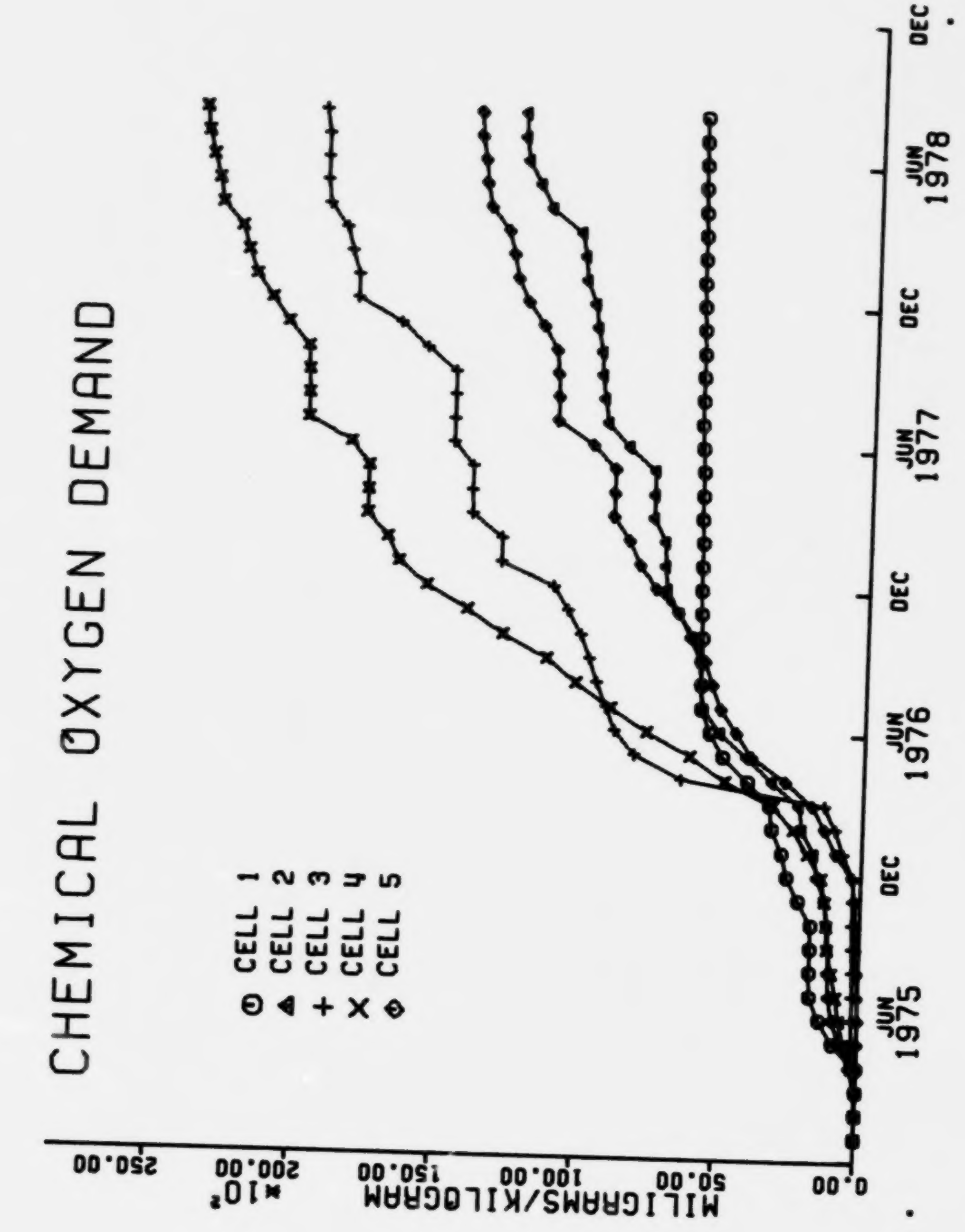

TOTAL SOLIDS
⊙ CELL 1
△ CELL 2
+ CELL 3
X CELL 4
◇ CELL 5
MILIGRAMS/KILOGRAM ×10²
0.00
40.00
80.00
120.00
160.00
200.00
CUMULATIVE LEACHATE VOLUME (LITERS) ×10²
0.00
12.50
25.00
37.50
50.00
62.50
75.00
87.50
100.00

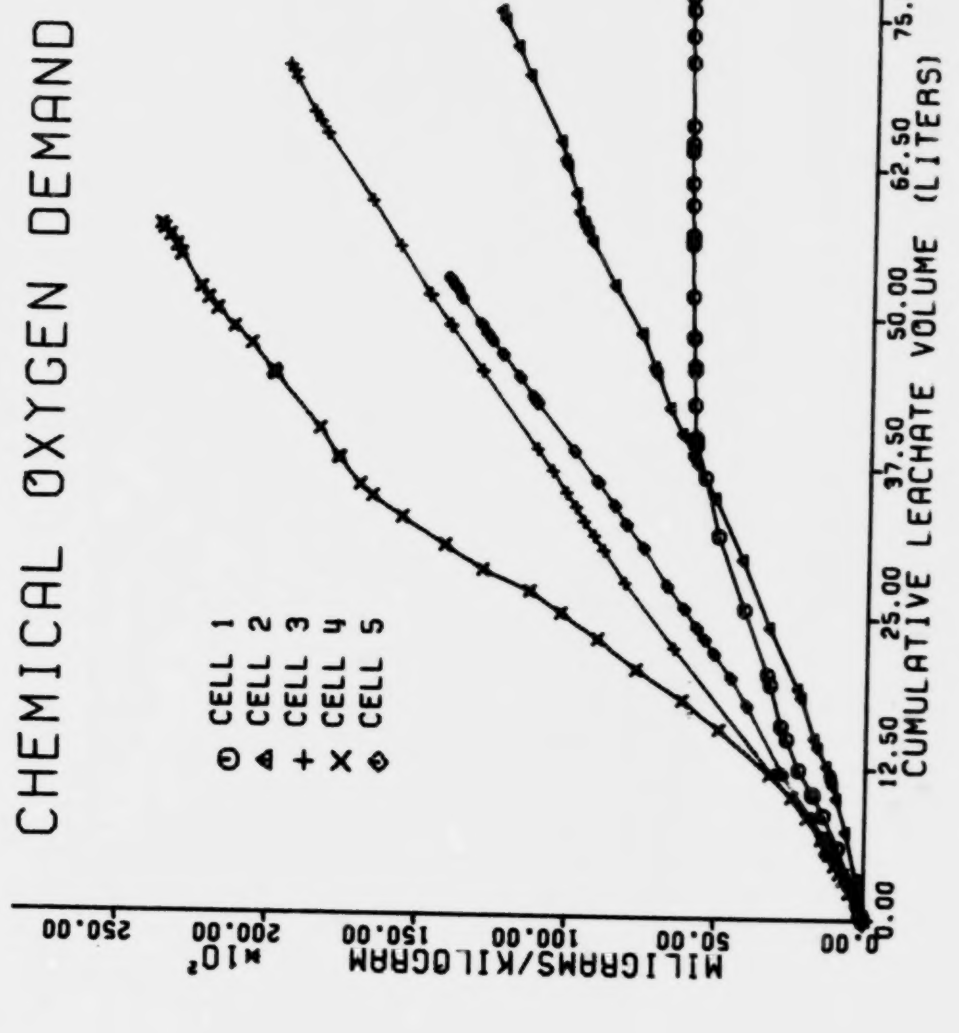

TOTAL ORGANIC CARBON
MILIGRAMS/KILOGRAM
×10²
0.00
20.00
40.00
60.00
80.00
100.00
⊙ CELL 1
▲ CELL 2
+ CELL 3
× CELL 4
◇ CELL 5
JUN 1975
DEC
JUN 1976
DEC
JUN 1977
DEC
JUN 1978
DEC

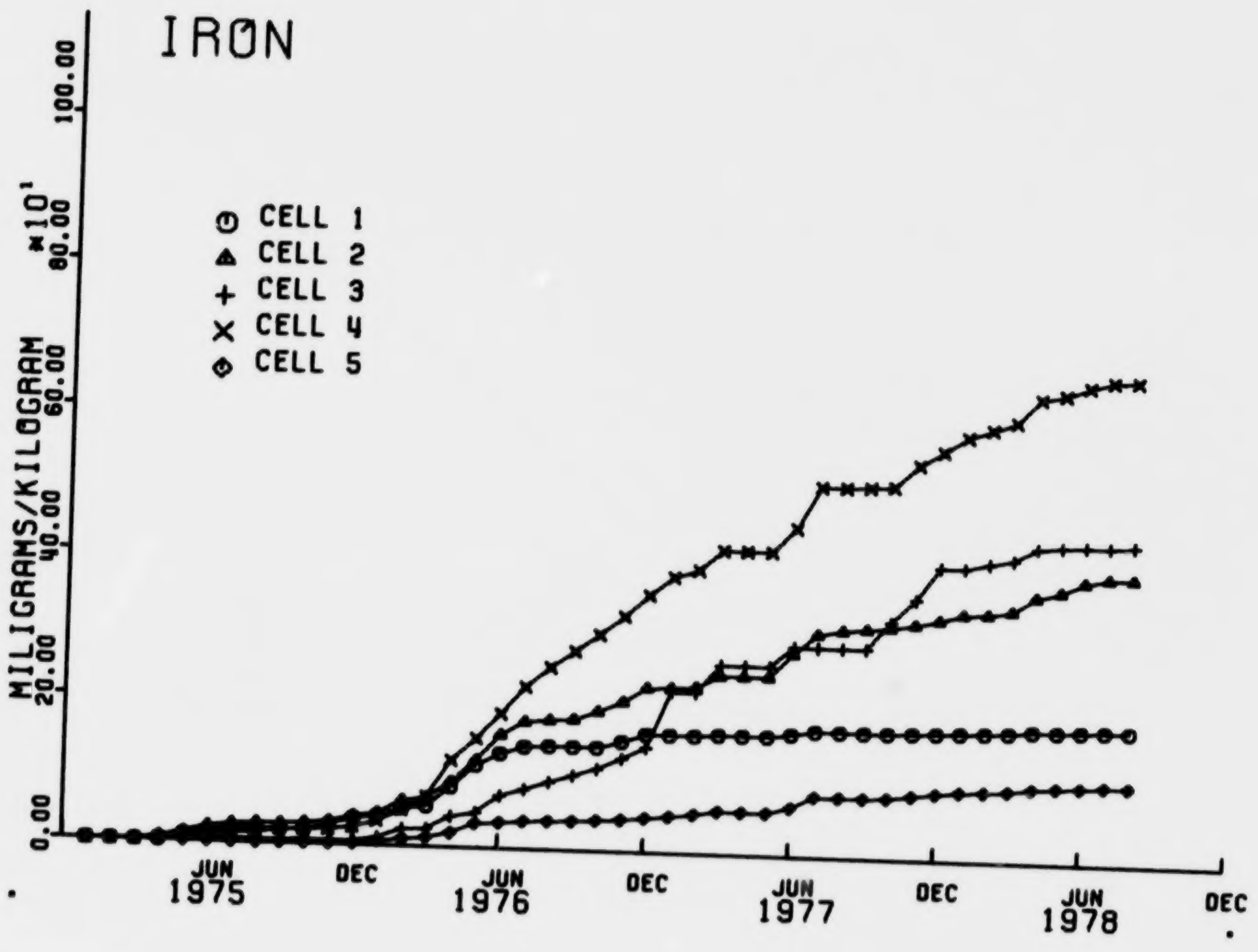

TOTAL ORGANIC CARBON
⊙ CELL 1
▲ CELL 2
+ CELL 3
X CELL 4
◇ CELL 5
MILIGRAMS/KILOGRAM ×10²
0.00
20.00
40.00
60.00
80.00
100.00
CUMULATIVE LEACHATE VOLUME (LITERS) ×10²
0.00
12.50
25.00
37.50
50.00
62.50
75.00
87.50
100.00

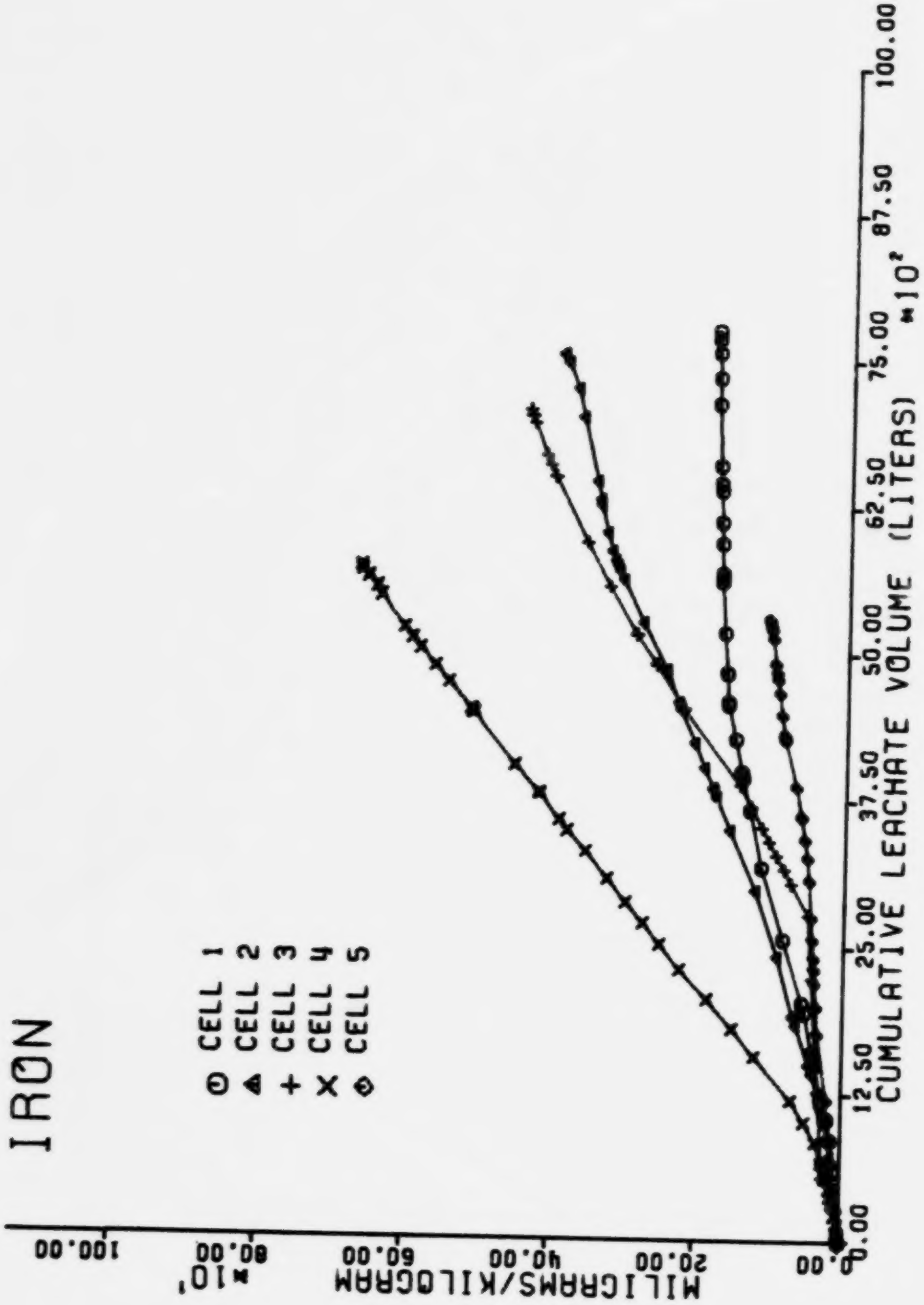

TØTAL KJELDAHL NITRØGEN
MILIGRAMS/KILØGRAM
0.00
80.00
160.00
240.00
320.00
400.00
JUN 1975
DEC
JUN 1976
DEC
JUN 1977
DEC
JUN 1978
DEC
⊙ CELL 1
▲ CELL 2
+ CELL 3
× CELL 4
◈ CELL 5

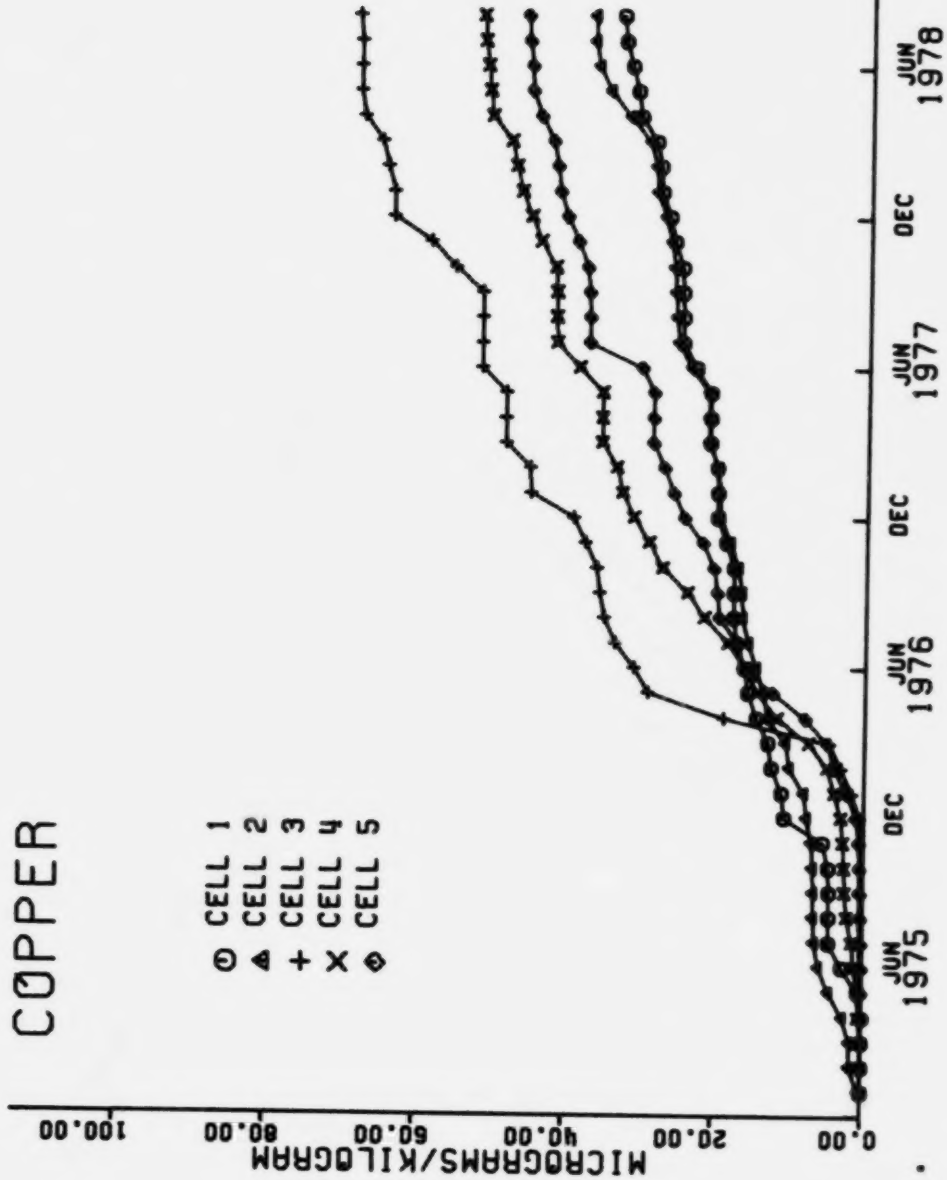

TOTAL KJELDAHL NITROGEN
⊖ CELL 1
△ CELL 2
+ CELL 3
X CELL 4
◇ CELL 5
MILIGRAMS/KILOGRAM
0.00
80.00
160.00
240.00
320.00
400.00
CUMULATIVE LEACHATE VOLUME (LITERS) ×10²
0.00
12.50
25.00
37.50
50.00
62.50
75.00
87.50
100.00

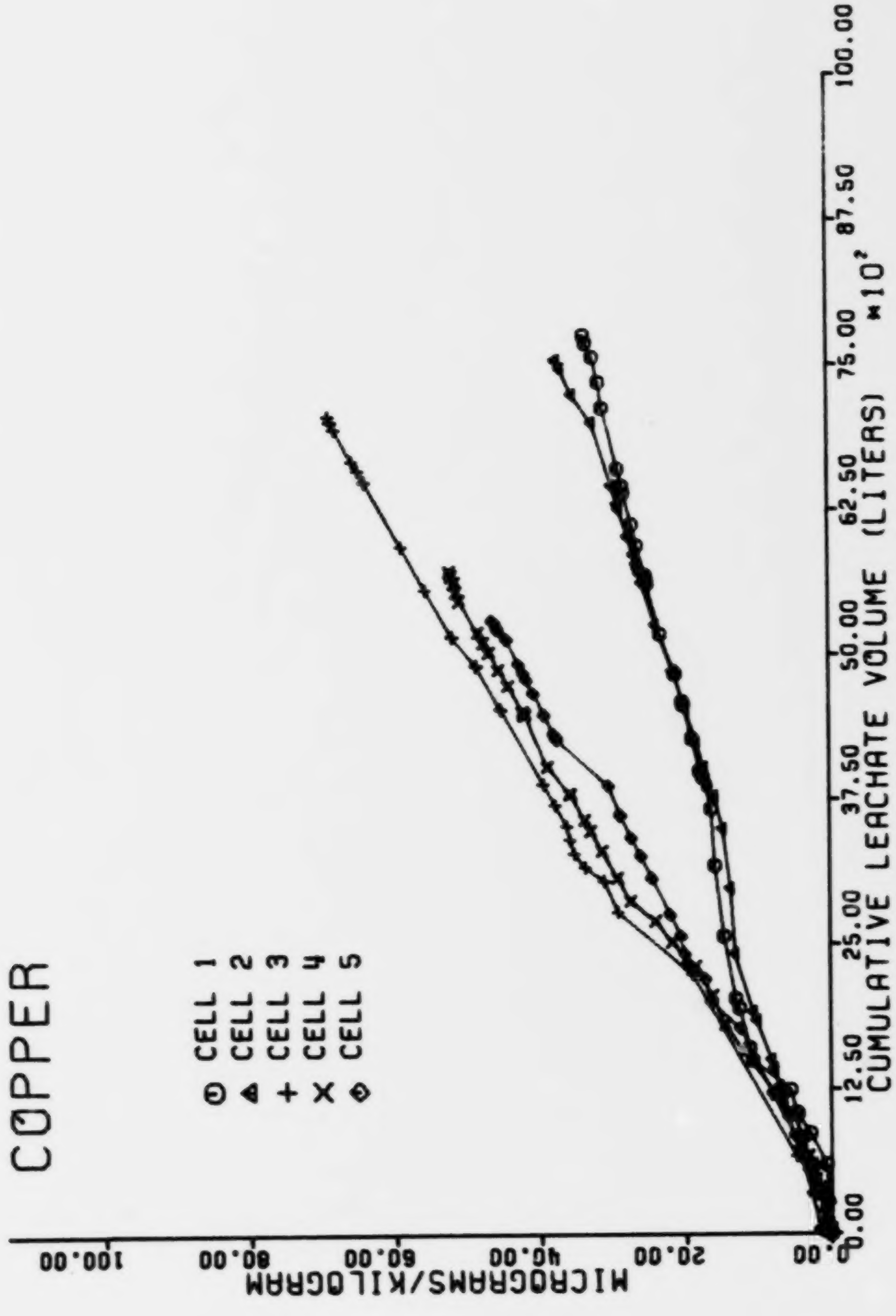

CHEMICAL OXYGEN DEMAND
⊙ CELL 1
▲ CELL 2
+ CELL 3
X CELL 4
◇ CELL 5
MILIGRAMS/KILOGRAM
*10²
0.00
50.00
100.00
150.00
200.00
250.00
JUN 1975
DEC
JUN 1976
DEC
JUN 1977
DEC
JUN 1978
DEC

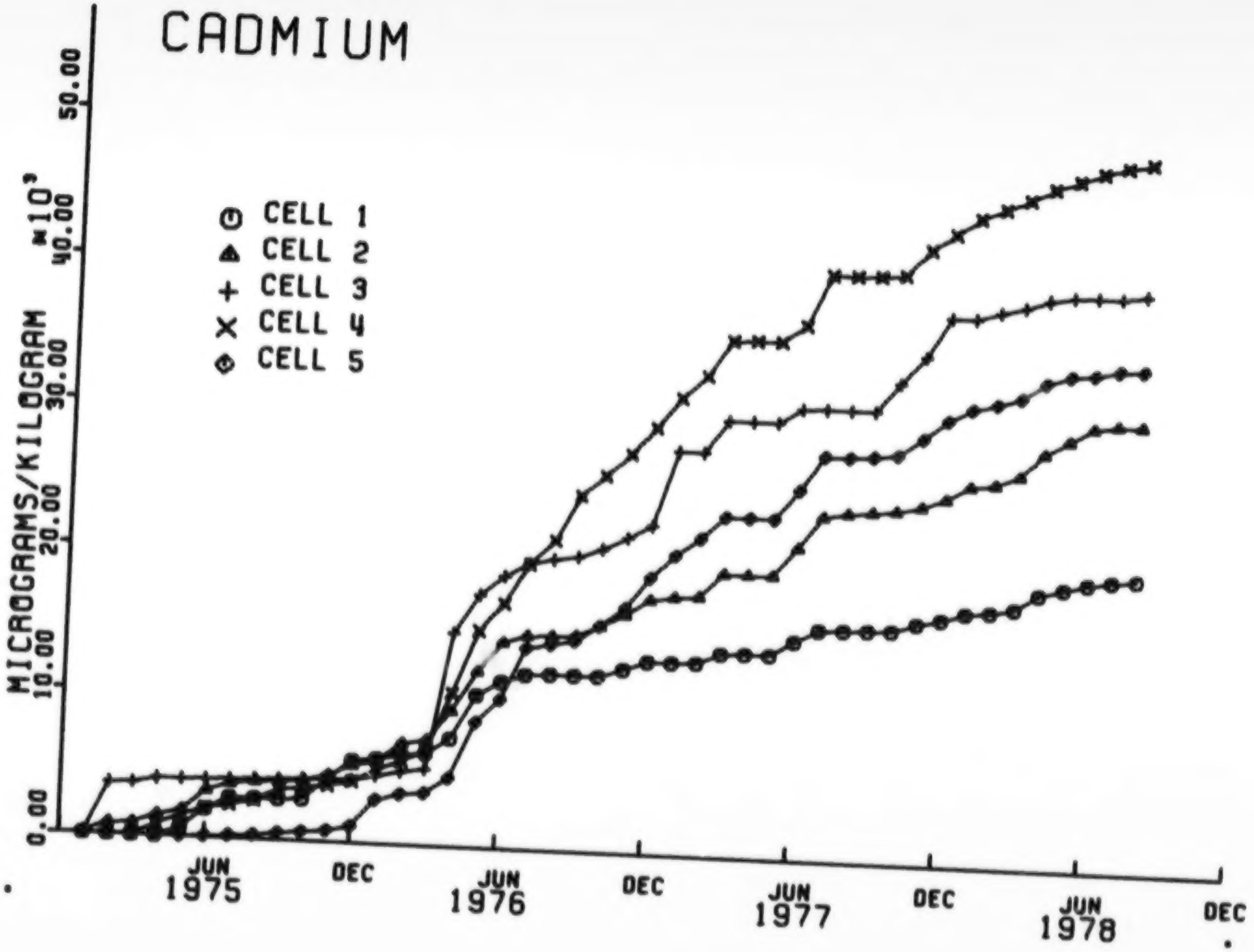

CHEMICAL OXYGEN DEMAND
⊙ CELL 1
△ CELL 2
+ CELL 3
× CELL 4
◇ CELL 5
MILIGRAMS/KILOGRAM
$\times 10^2$
0.00
50.00
100.00
150.00
200.00
250.00
CUMULATIVE LEACHATE VOLUME (LITERS) $\times 10^2$
0.00
12.50
25.00
37.50
50.00
62.50
75.00
87.50
100.00

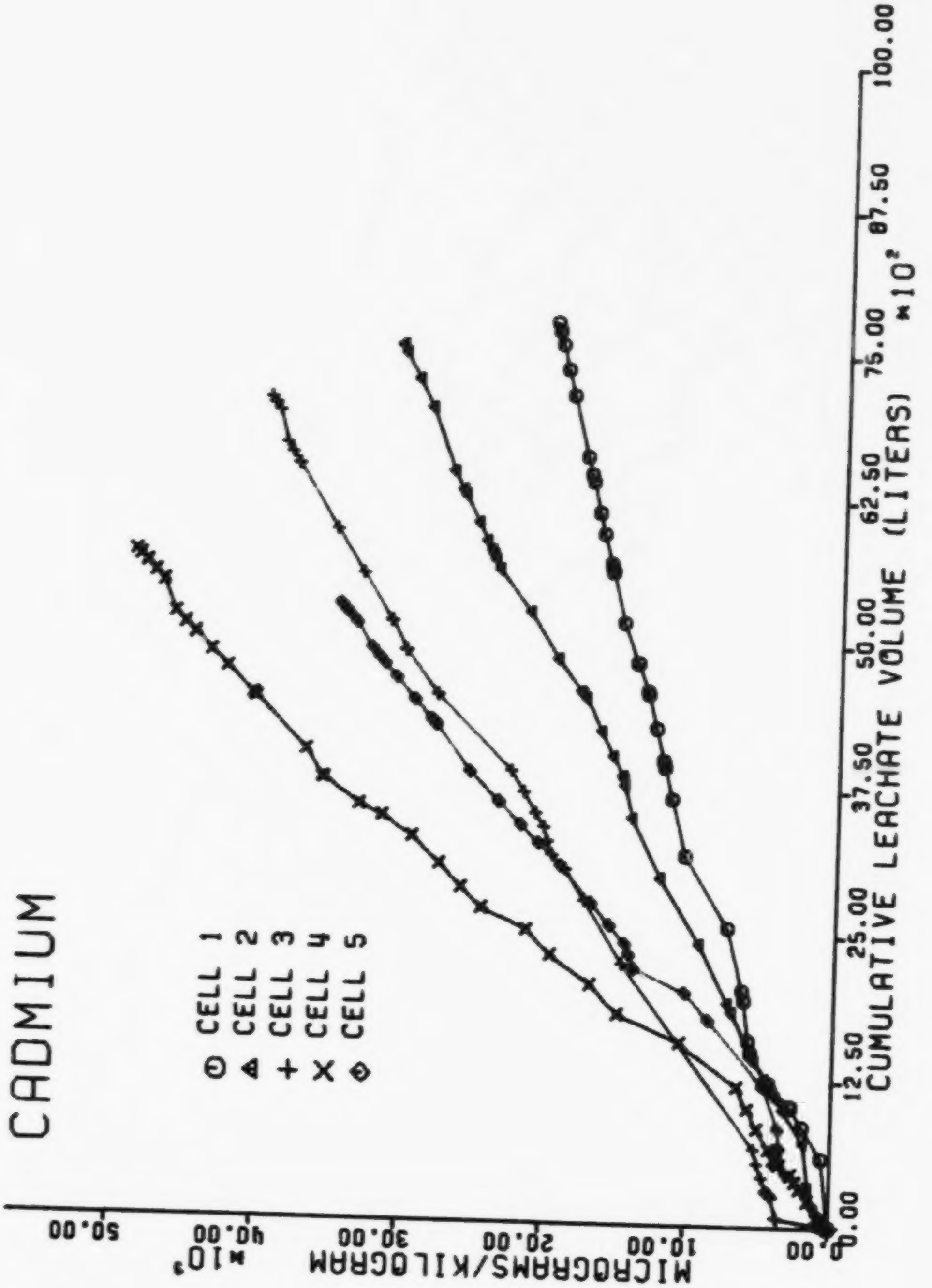

IRON
CELL 1
CELL 2
CELL 3
CELL 4
CELL 5
MILIGRAMS/KILOGRAM
$\times 10^1$
0.00
20.00
40.00
60.00
80.00
100.00
JUN 1975
DEC
JUN 1976
DEC
JUN 1977
DEC
JUN 1978
DEC

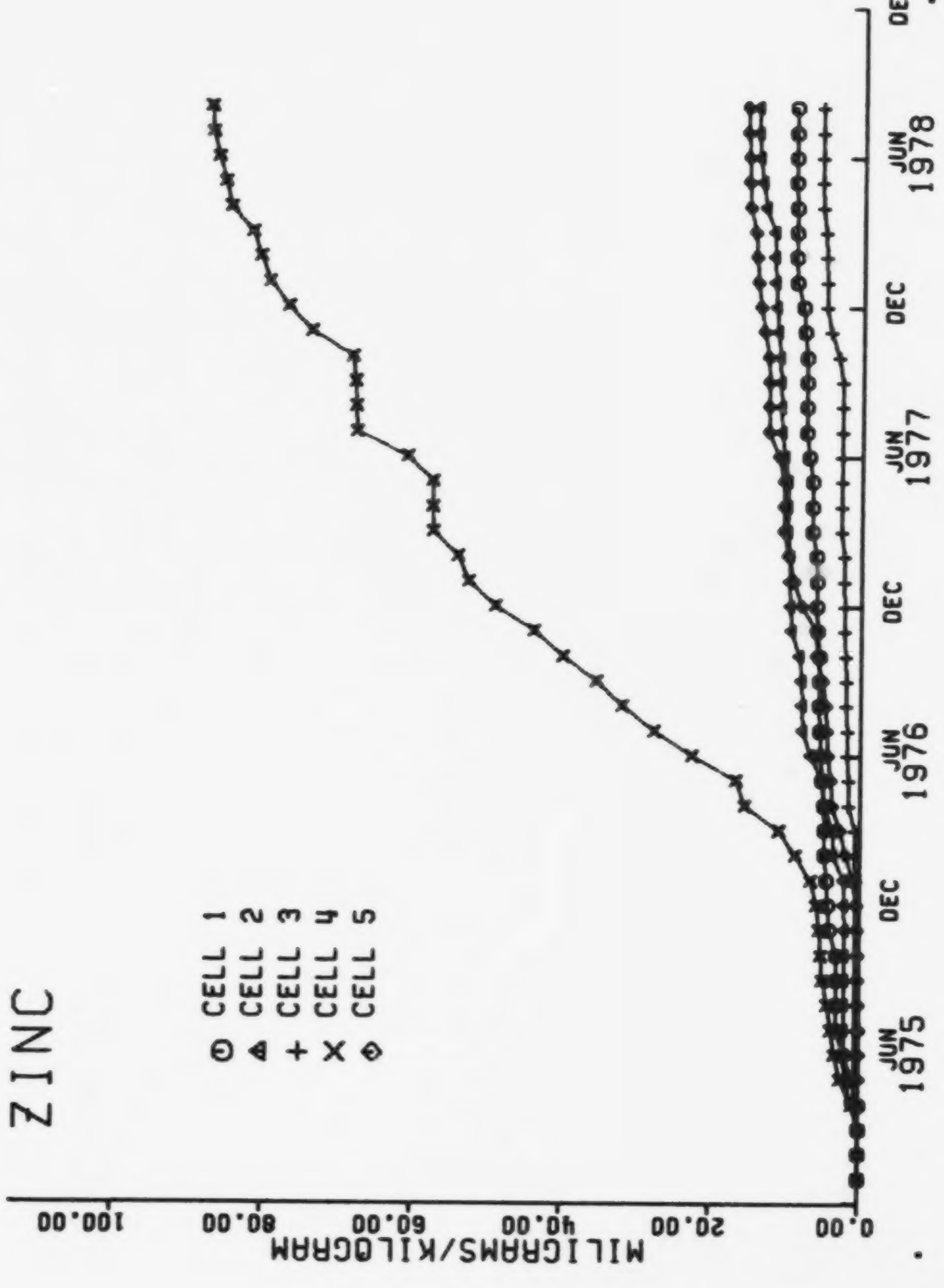

IRON
CELL 1
CELL 2
CELL 3
CELL 4
CELL 5
MILIGRAMS/KILOGRAM
*10¹
0.00
20.00
40.00
60.00
80.00
100.00
CUMULATIVE LEACHATE VOLUME (LITERS) *10²
0.00
12.50
25.00
37.50
50.00
62.50
75.00
87.50
100.00

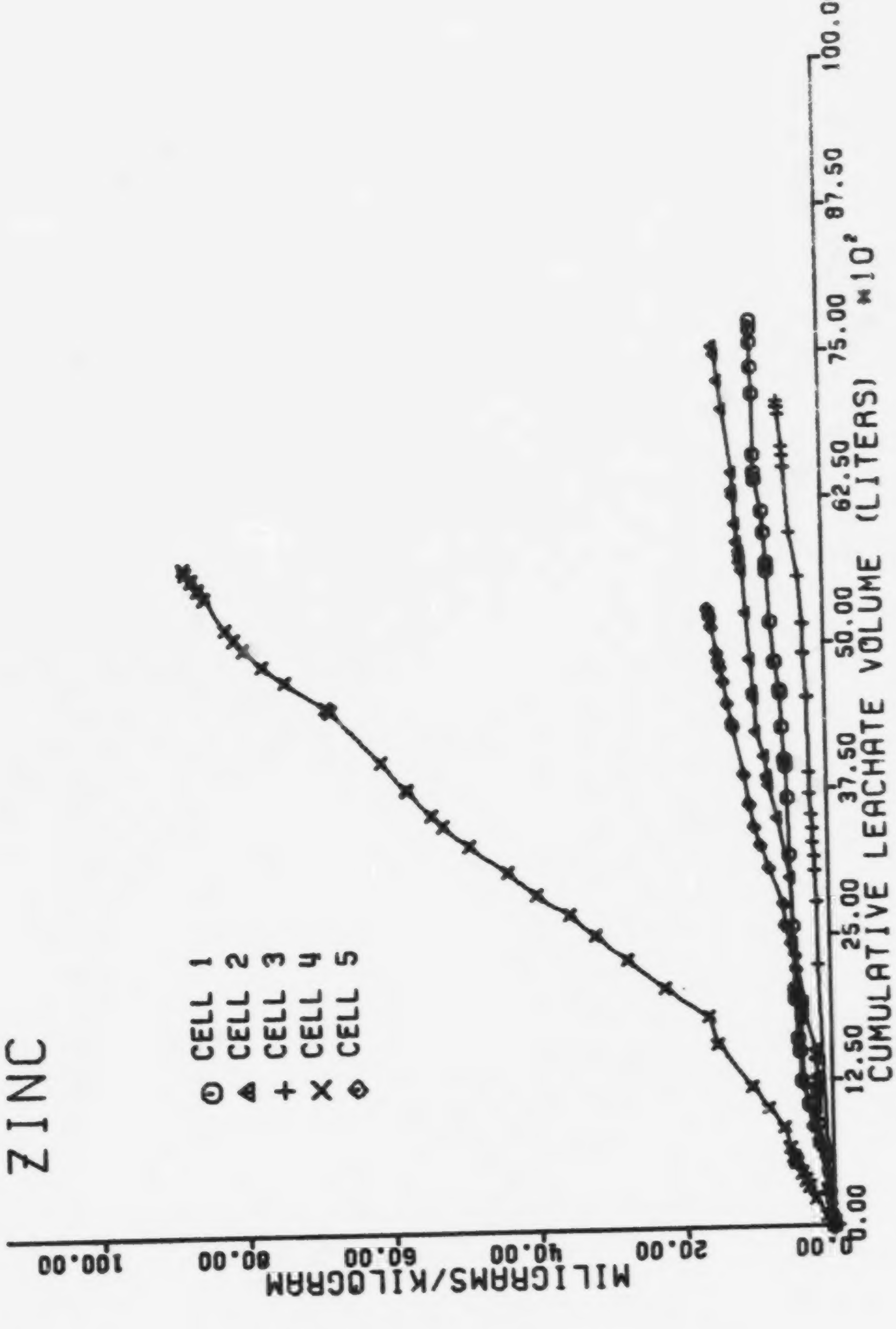

COPPER
⊙ CELL 1
▲ CELL 2
+ CELL 3
X CELL 4
◆ CELL 5
MICROGRAMS/KILOGRAM
0.00
20.00
40.00
60.00
80.00
100.00
JUN 1975
DEC
JUN 1976
DEC
JUN 1977
DEC
JUN 1978
DEC

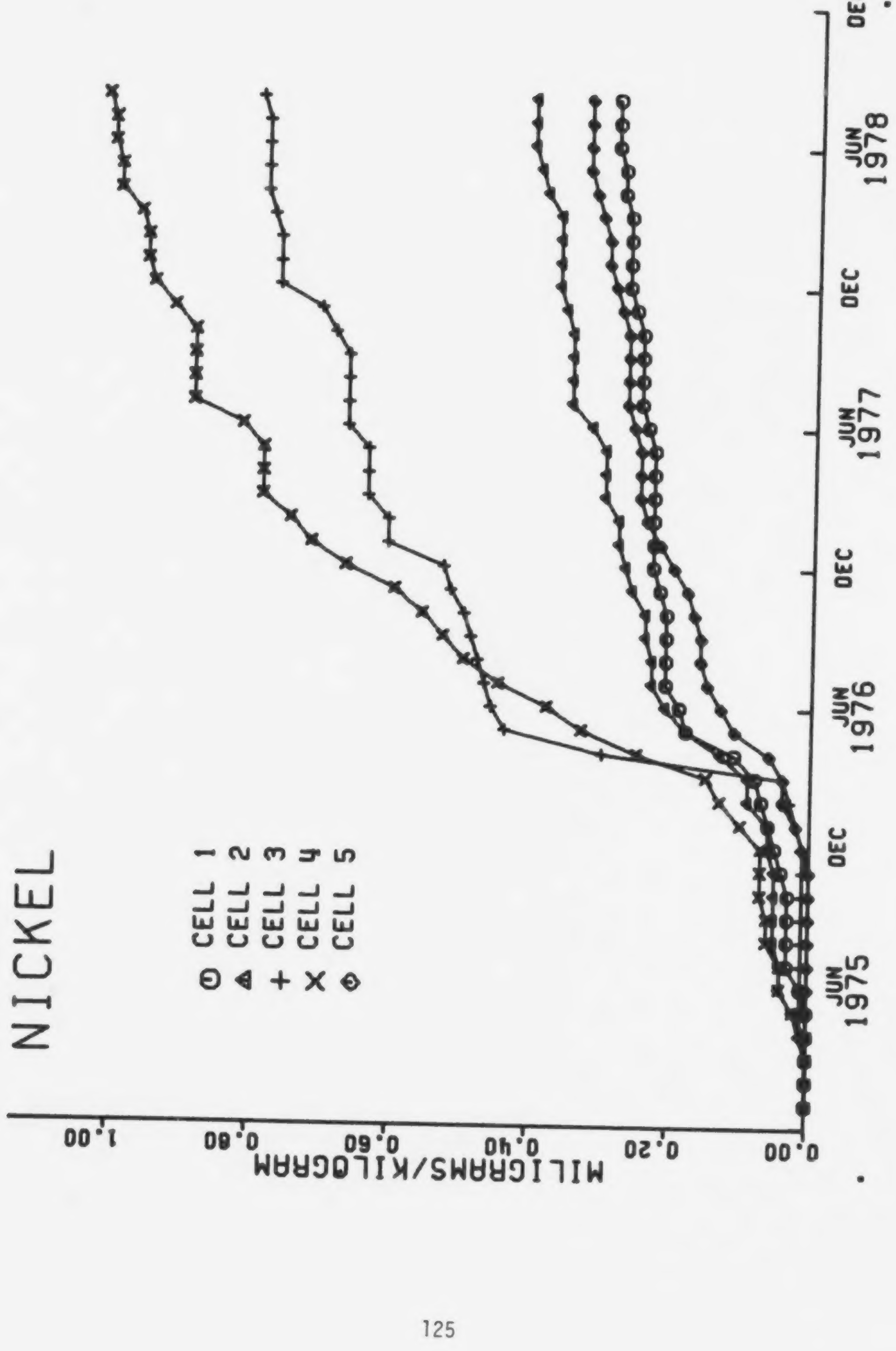

COPPER
⊖ CELL 1
▲ CELL 2
+ CELL 3
X CELL 4
◇ CELL 5
MICROGRAMS/KILOGRAM
0.00
20.00
40.00
60.00
80.00
100.00
CUMULATIVE LEACHATE VOLUME (LITERS) ×10²
0.00
12.50
25.00
37.50
50.00
62.50
75.00
87.50
100.00

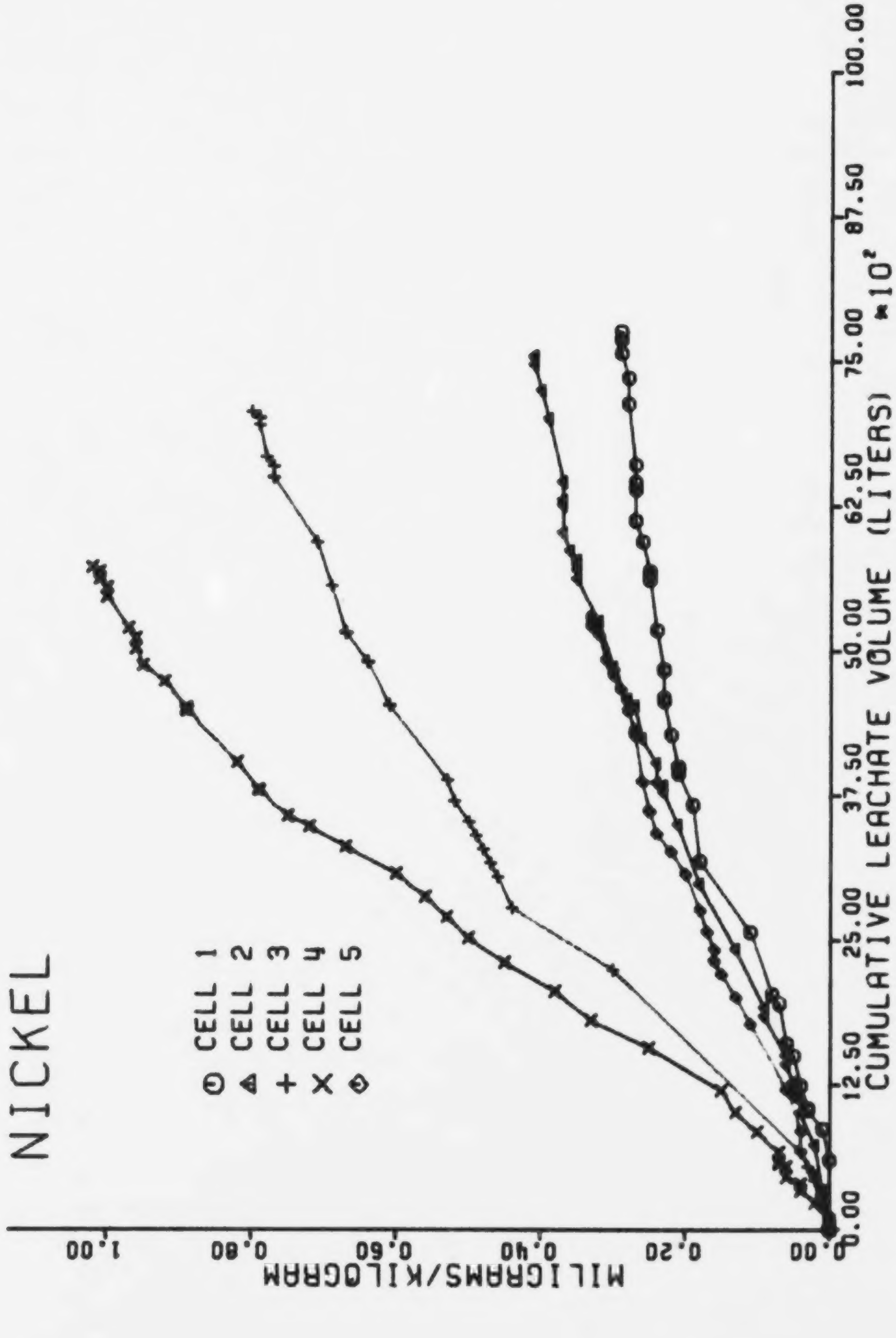

CADMIUM
CELL 1
CELL 2
CELL 3
CELL 4
CELL 5
MICROGRAMS/KILOGRAM
*10³
0.00
10.00
20.00
30.00
40.00
50.00
JUN 1975
DEC
JUN 1976
DEC
JUN 1977
DEC
JUN 1978
DEC

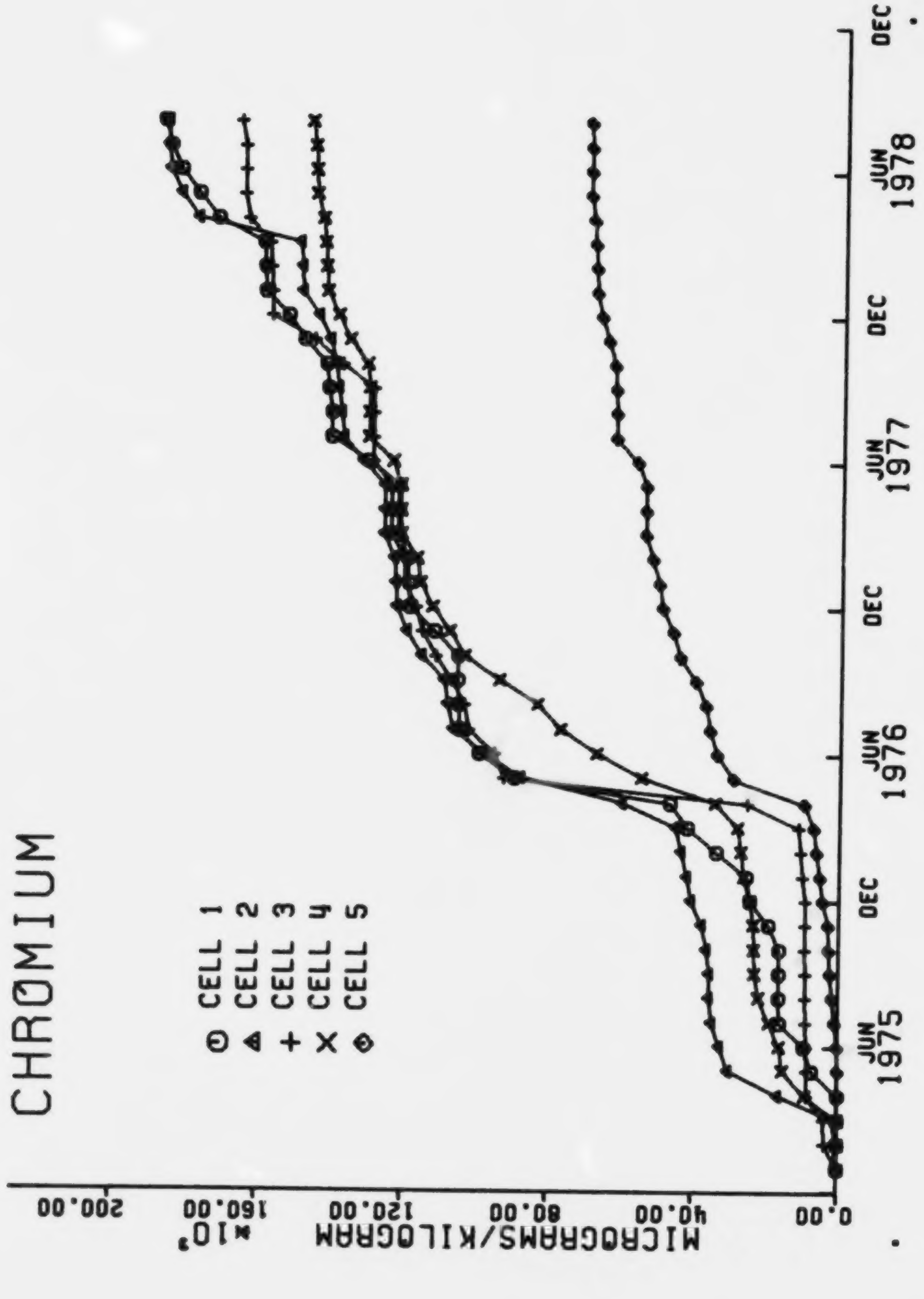

CADMIUM
CELL 1
CELL 2
CELL 3
CELL 4
CELL 5
MICROGRAMS/KILOGRAM ×10³
0.00
10.00
20.00
30.00
40.00
50.00
CUMULATIVE LEACHATE VOLUME (LITERS) ×10²
0.00
12.50
25.00
37.50
50.00
62.50
75.00
87.50
100.00

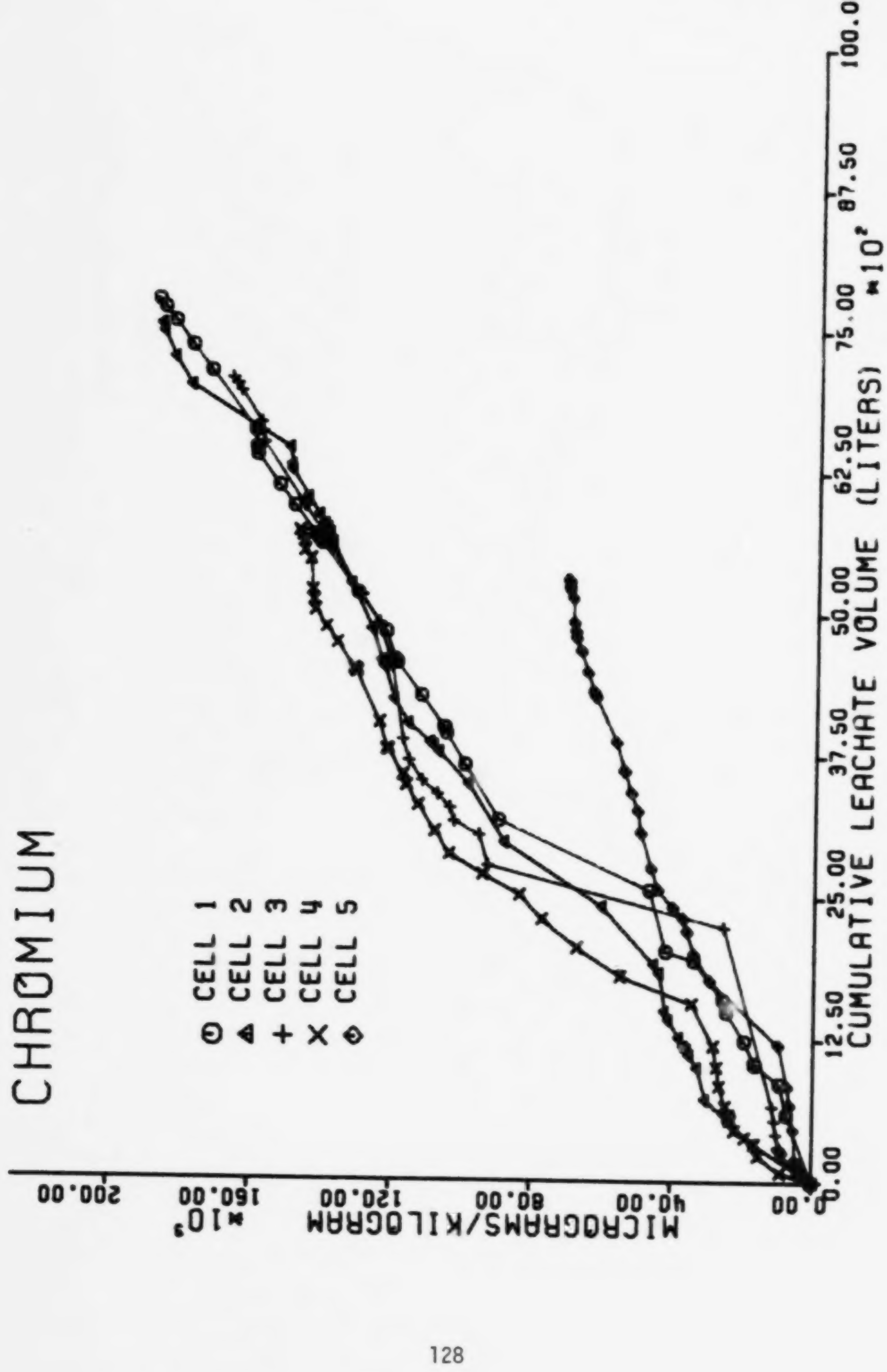

ZINC
⊖ CELL 1
▲ CELL 2
+ CELL 3
X CELL 4
◇ CELL 5
MILIGRAMS/KILOGRAM
0.00
20.00
40.00
60.00
80.00
100.00
JUN 1975
DEC
JUN 1976
DEC
JUN 1977
DEC
JUN 1978
DEC

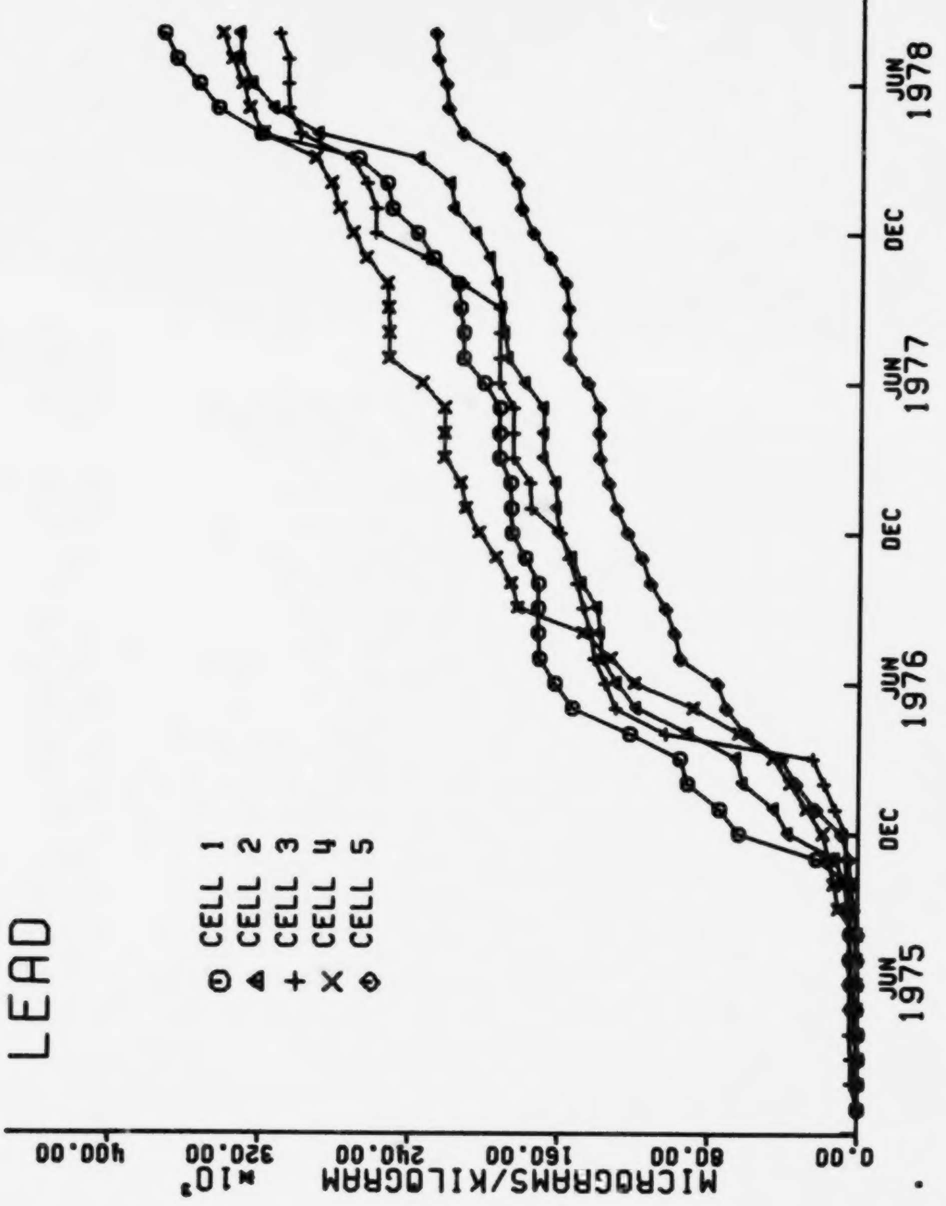

ZINC
⊙ CELL 1
△ CELL 2
+ CELL 3
X CELL 4
◇ CELL 5
MILIGRAMS/KILOGRAM
0.00
20.00
40.00
60.00
80.00
100.00
CUMULATIVE LEACHATE VOLUME (LITERS) *10²
0.00
12.50
25.00
37.50
50.00
62.50
75.00
87.50
100.00

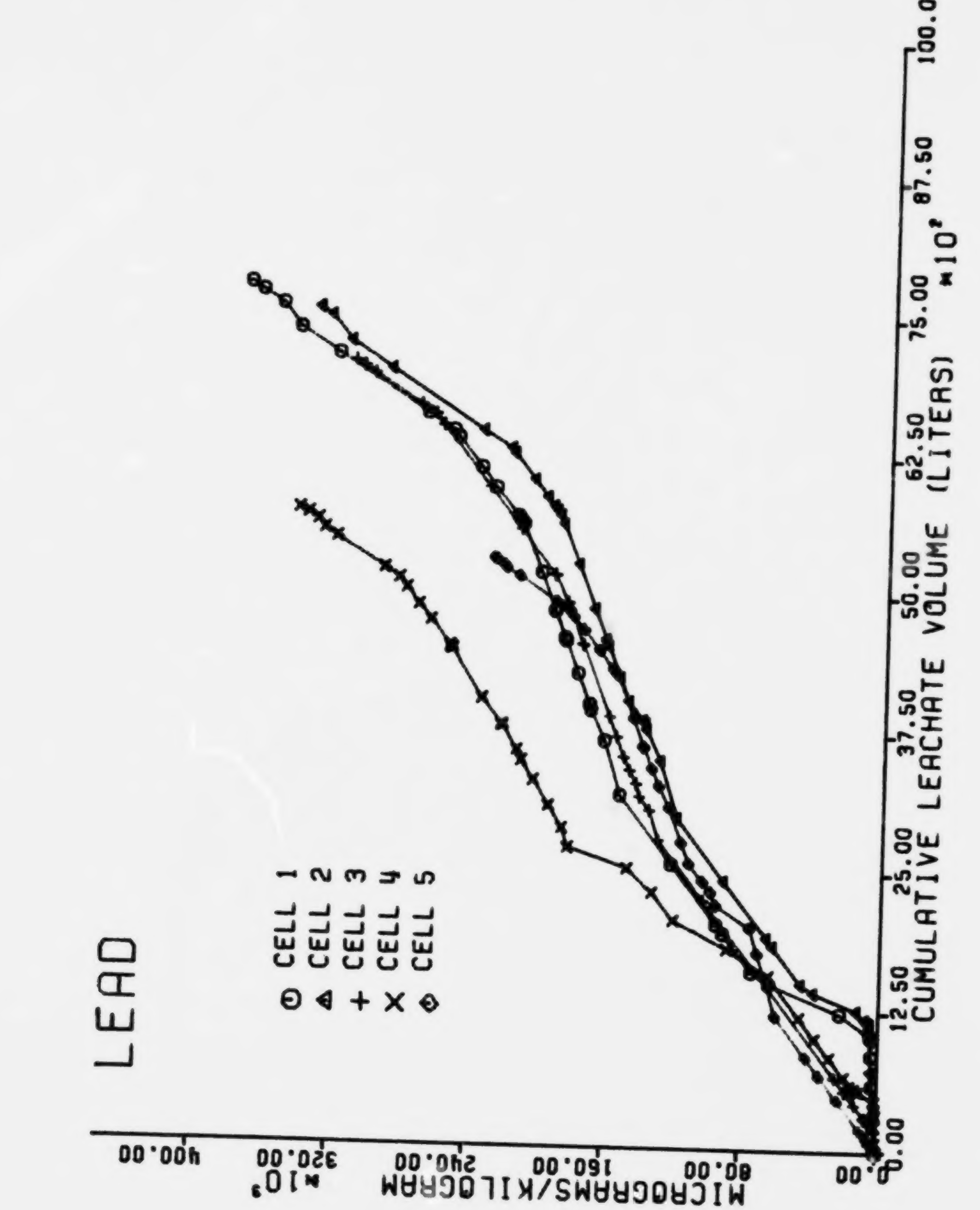

NICKEL
CELL 1
CELL 2
CELL 3
CELL 4
CELL 5
MILIGRAMS/KILOGRAM
0.00
0.20
0.40
0.60
0.80
1.00
JUN 1975
DEC
JUN 1976
DEC
JUN 1977
DEC
JUN 1978
DEC

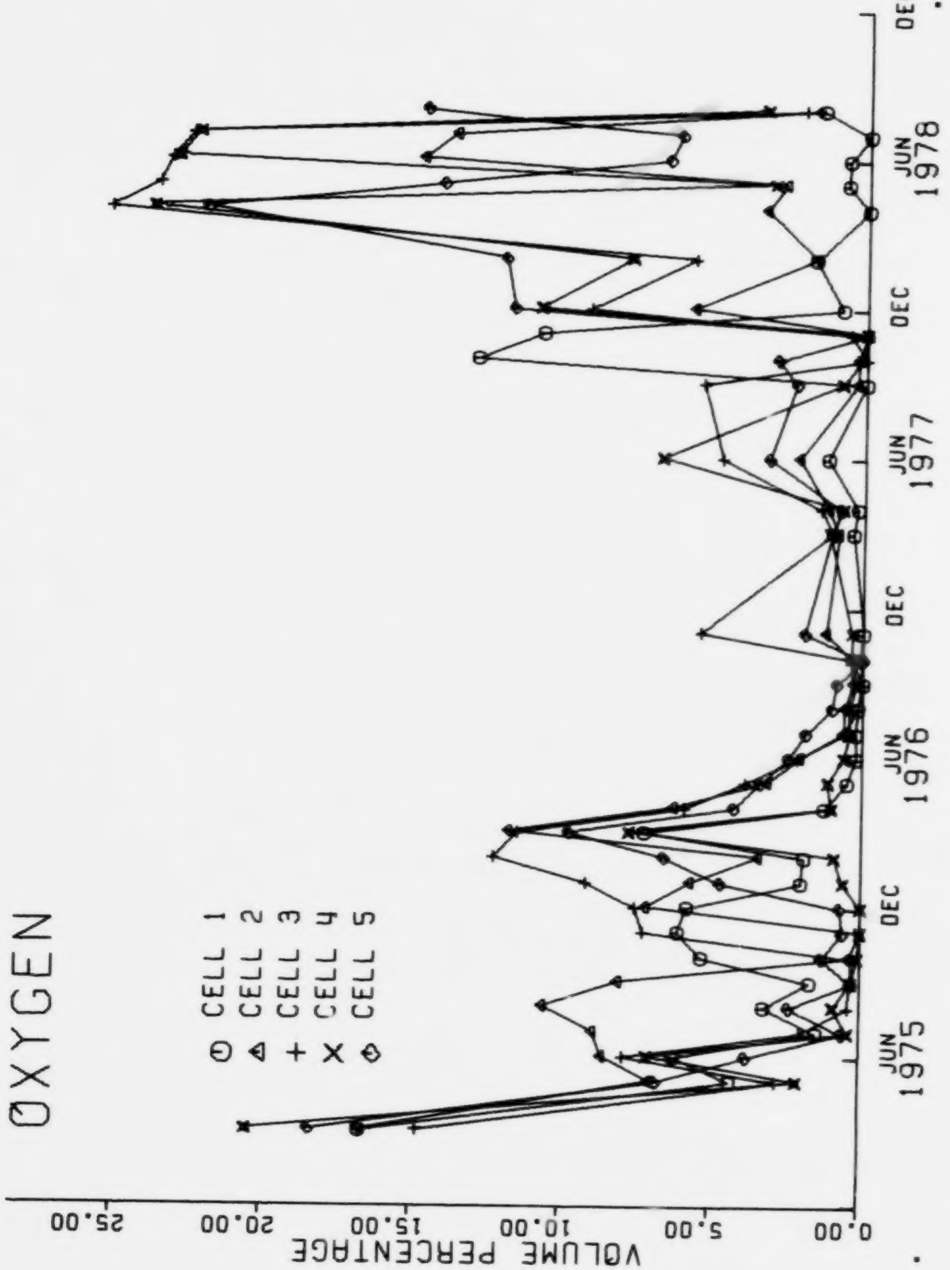

NICKEL
⊙ CELL 1
△ CELL 2
+ CELL 3
× CELL 4
◇ CELL 5
MILIGRAMS/KILOGRAM
0.00
0.20
0.40
0.60
0.80
1.00
0.00
12.50
25.00
37.50
50.00
62.50
75.00
87.50
100.00
CUMULATIVE LEACHATE VOLUME (LITERS) $*10^2$

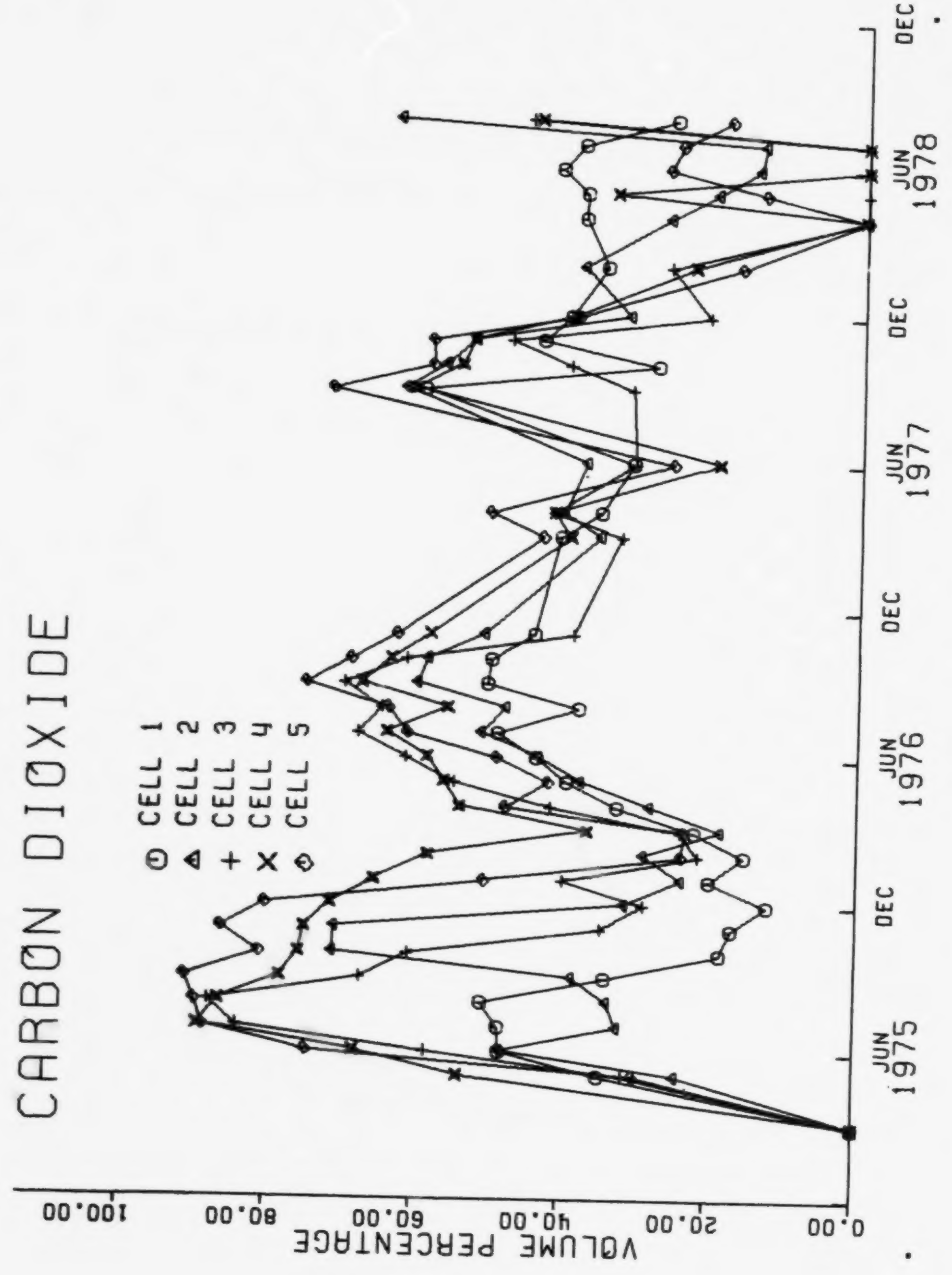

CHROMIUM
⊙ CELL 1
▲ CELL 2
+ CELL 3
X CELL 4
◇ CELL 5
MICROGRAMS/KILOGRAM
×10³
0.00
40.00
80.00
120.00
160.00
200.00
JUN
1975
DEC
JUN
1976
DEC
JUN
1977
DEC
JUN
1978
DEC

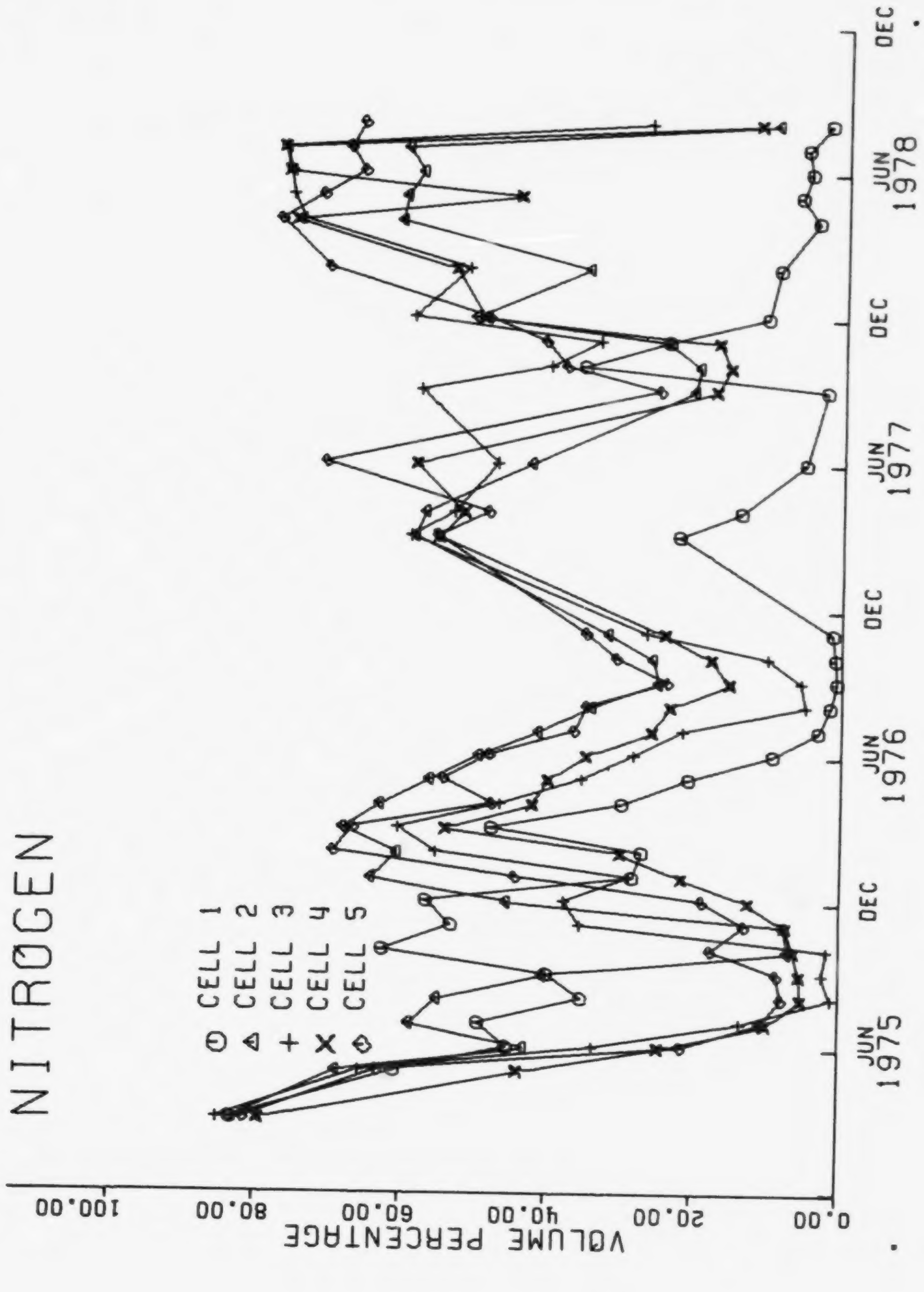

CHROMIUM
⊙ CELL 1
▲ CELL 2
+ CELL 3
X CELL 4
◇ CELL 5
MICROGRAMS/KILOGRAM ×10³
0.00
40.00
80.00
120.00
160.00
200.00
CUMULATIVE LEACHATE VOLUME (LITERS) ×10²
0.00
12.50
25.00
37.50
50.00
62.50
75.00
87.50
100.00

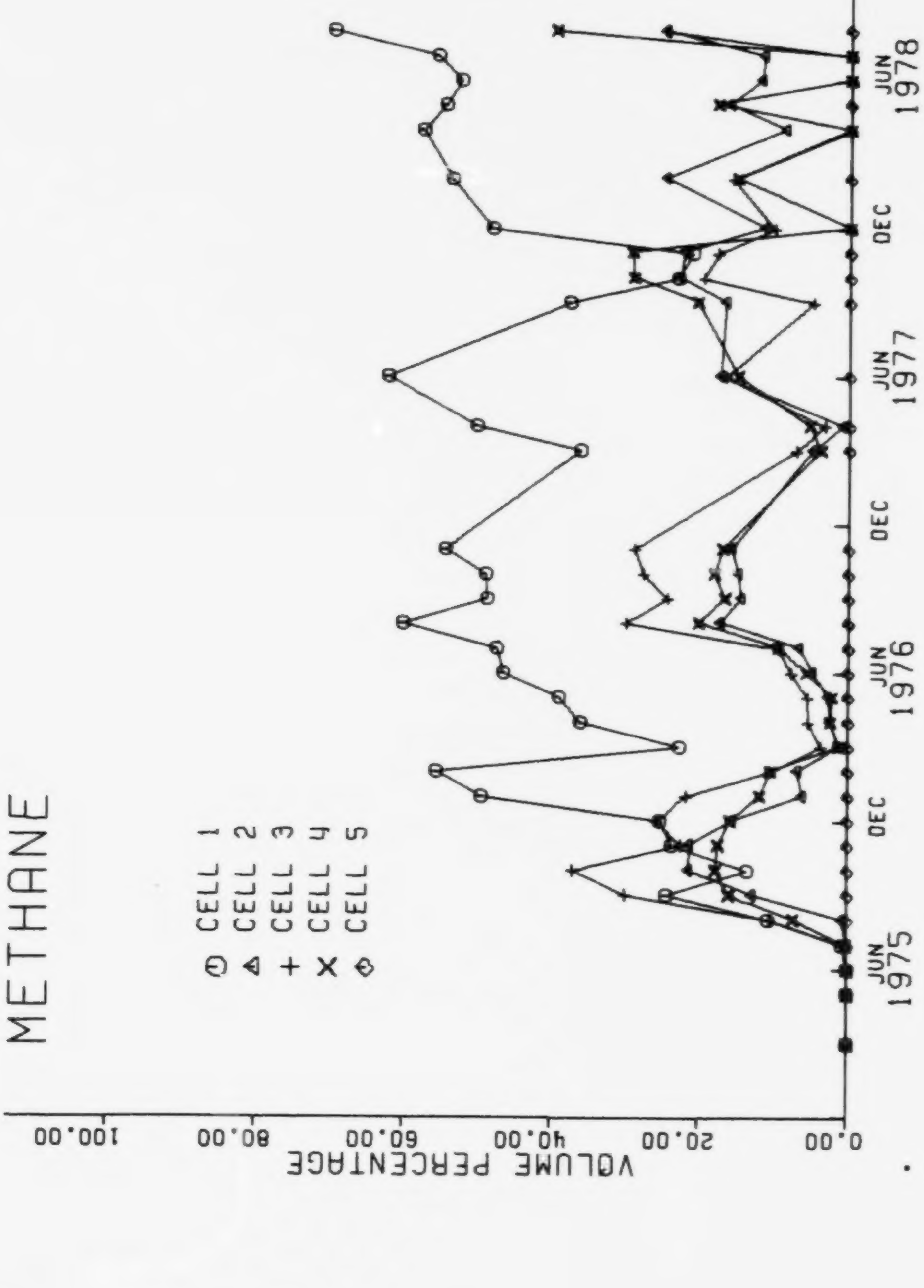

LEAD
⊙ CELL 1
▲ CELL 2
+ CELL 3
X CELL 4
◇ CELL 5
MICROGRAMS/KILOGRAM
*10³
0.00
80.00
160.00
240.00
320.00
400.00
JUN 1975
DEC
JUN 1976
DEC
JUN 1977
DEC
JUN 1978
DEC


LEAD
⊙ CELL 1
△ CELL 2
+ CELL 3
× CELL 4
◇ CELL 5
MICROGRAMS/KILOGRAM ×10³
0.00
80.00
160.00
240.00
320.00
400.00
CUMULATIVE LEACHATE VOLUME (LITERS) ×10²
0.00
12.50
25.00
37.50
50.00
62.50
75.00
87.50
100.00


ØXYGEN
CELL 1
CELL 2
CELL 3
CELL 4
CELL 5
VØLUME PERCENTAGE
0.00
5.00
10.00
15.00
20.00
25.00
JUN 1975
DEC
JUN 1976
DEC
JUN 1977
DEC
JUN 1978
DEC


CARBON DIOXIDE
⊖ CELL 1
△ CELL 2
+ CELL 3
X CELL 4
◇ CELL 5
VOLUME PERCENTAGE
0.00
20.00
40.00
60.00
80.00
100.00
JUN 1975
DEC
JUN 1976
DEC
JUN 1977
DEC
JUN 1978
DEC


NITRØGEN
CELL 1
CELL 2
CELL 3
CELL 4
CELL 5
VØLUME PERCENTAGE
0.00
20.00
40.00
60.00
80.00
100.00
JUN
1975
DEC
JUN
1976
DEC
JUN
1977
DEC
JUN
1978
DEC


METHANE
CELL 1
CELL 2
CELL 3
CELL 4
CELL 5
VOLUME PERCENTAGE
0.00
20.00
40.00
60.00
80.00
100.00
JUN 1975
DEC
JUN 1976
DEC
JUN 1977
DEC
JUN 1978
DEC

WATER ADDITION SCHEDULE

Month	Water Applied		
	cm	gal	Liter
January	0	0	0
February	5.54	104.5	396
March	11.08	209.0	791
April	11.08	209.0	791
May	11.08	209.0	791
June	5.54	104.5	396
July	0	0	0
August	0	0	0
September	0	0	0
October	5.54	104.5	396
November	5.54	104.5	396
December	5.54	104.5	396

WATER BALANCE SUMMARY

Test Cell	Time to Reach F.C.* in Mths.	Water Added to Reach F.C. (ℓ)	Avg. Moisture Retained in Cell Subsequent to Reaching F.C. (ℓ)	Avg. Moisture (ml/kg Refuse)
1	5	2397	$\bar{x}$ = 3419 σ ± 1226 (36%)	$\bar{x}$ = 301 σ ± 95 (32%)
2	2	396	$\bar{x}$ = 3494 σ ± 1350 (39%)	$\bar{x}$ = 300 σ ± 116 (39%)
4	5	1397	$\bar{x}$ = 5038 σ ± 1725 (34%)	$\bar{x}$ = 465 σ ± 159 (34%)
5	6	2816	$\bar{x}$ = 5733 σ ± 1728 (30%)	$\bar{x}$ = 590 σ ± 178 (30%)

* Field capacity is defined as the maximum moisture content which a solid (or solid waste) can retain in a gravitational field without producing continuous downward percolation (Reference 2). Data is presented in terms of "field capacity" meaning the initiation of continuous leachate production.

COMPARISON OF CUMULATIVE MASSES OF LEACHATE PARAMETERS ON A BASIS OF EQUAL LEACHATE VOLUME

Parameter	Cell 1 Baled/Shredded (mg/kg)	Cell 2 Baled (mg/kg)	Cell 3 Baled/Constant Moisture (mg/kg)	Cell 4 Shredded (mg/kg)	Cell 5 Unprocessed (mg/kg)
Total Alkalinity	2411	2299	3929	6472	3421
Total Solids	5600	5729	9592	17742	9154
TOC	3190	3561	5746	8833	5452
TKN	94	119	187	335	230
COD	6030	8494	14926	22492	13887
Iron	170	275	293	608	104
Copper	23	23	52	48	45
Cadmium	15.14	21.26	31.37	46.15	38.99
Zinc	6475	6266	2240	50336	11465
Nickel	0.24	0.32	0.67	0.97	0.33
Chromium	131.0	131.3	130.4	143.5	70.07
Lead	202.3	178.2	195.4	292.9	219.8

PATHOGEN CONTENT OF LANDFILL LEACHATE

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THE UNIVERSITY OF CINCINNATI
Cincinnati, Ohio 45221

and

D. BRUNNER

SOLID AND HAZARDOUS WASTE RESEARCH DIVISION
MUNICIPAL ENVIRONMENTAL RESEARCH LABORATORY
U. S. ENVIRONMENTAL PROTECTION AGENCY
Cincinnati, Ohio 45268

ABSTRACT

Increasing amounts of municipal solid waste, sewage sludge, hospital waste, chemicals, etc., are being sent to sanitary landfills and open dumps. This material may contain pathogenic microorganisms which may be leached out with rain water, and subsequently contaminate ground and surface waters. It was found in our studies that older landfills contained Bacillus sp., Streptococcus sp., Corynebacterium sp., and yeast. Pathogens found in these older landfills included Listeria monocytogenes, Acinetobacter sp., Moraxella sp., and Allescheria boydii. Fecal streptococcal levels in all these landfills studied were high. The fecal streptococcal MPN test was not always specific, since in some leachates Gram positive rods, in addition to the streptococci, gave purple growth in ethyl violet azide broth. The leachate also showed toxicity when added to pure cultures of Gram negative rods and streptococci by repressing plate counts by at least two logs.

Leachates from six experimental laboratory lysimeters at the University of Cincinnati containing various combinations of municipal and hospital wastes and sewage sludge were also studied. Eighty-five species of streptococci were isolated from these experimental lysimeters, and fifty-seven species or 67 percent were enterococci. This indicated that fecal material was present in the lysimeter contents. Pathogenic isolations have not yet been made in these ongoing laboratory studies.

Leachates may present a microbial health hazard since fecal indicator bacterial forms such as fecal streptococci are found in them. However, only a limited number of pathogens have been isolated from older landfill leachates and none have been isolated as yet from the ongoing University of Cincinnati experimental lysimeters.

INTRODUCTION

Today one of our primary concerns is the assessment of the environmental health hazard associated with the disposal of solid wastes in sanitary landfills or open dumps. These solid wastes contain untreated animal and human fecal matter, animal remains, and sewage sludge. The use of disposable diapers is one way human excreta becomes present in solid waste. Ground and surface waters may become contaminated by microbial movement from the landfill site through the surrounding soil by the leaching action of rainwater or by the addition of liquid wastes (Peterson, 1971, 1974; Brunner, 1974; Engelbrecht et al., 1974). Human health could thus be affected through direct contact with the solid waste, its leachate or indirectly through contaminated water, air, land, or via biological vectors. The potential hazard from pathogenic agents reaching sanitary landfills is a function of three interrelated conditions (Engelbrecht et al., 1974):

1. the density and nature of the pathogens initially placed in the landfill,
2. the ability of the pathogens to survive or to retain their infectious properties in the landfill environment or in the leachate, and
3. the ability of the pathogens to move through the landfill with the leachate into the surrounding environment and present a hazard to human health.

As pointed out by Engelbrecht et al. (1974), Pohland and Engelbrecht (1976), and by Blannon and Peterson (1974), specific information on the survival of infectious agents in sanitary landfills is essentially nonexistent. It is thus the prime concern of these studies to characterize and evaluate the persistence of microbes, especially those of pathogenic significance or indication, in landfill leachates.

EXPERIMENTAL APPROACH

The objectives of this study were to evaluate the microbial health hazard associated with leachate. The initial and continuing effort of Phase I was to determine periodically the presence of microbial (bacterial and fungal) pathogens in leachates obtained from several existing commercial and experimental landfills, and to determine the species of fecal streptococci commonly found in leachate in order to ascertain their significance. The Phase II study was undertaken to study the microbial populations emanating from recently deposited wastes.

The sources for leachate studied under Phase I are briefly described in Table 1. The commercial landfill, currently operational, covers approximately 700 acres and was begun in 1954. Leachate is collected from a pipe which drains the landfill leachate from a filled-in 15 acre lake, assumed to be the lowest drainage area for the landfill. The leachate is assumed to be representative of the entire landfill and is at least seasonally diluted by groundwater discharge. The site receives municipal and commercial wastes and had accepted industrial wastes in the past.

The Walton, Kentucky sources are from the U.S. EPA (SHWRD) Boone County Field Site. The 435 ton landfill was constructed in two weeks during June, 1971, with municipal refuse and covered with a clay loam placed on a 2 percent slope. Leachate has been volumetrically collected since August, 1971. The 2 ton landfill was also constructed of municipal refuse, out filled and placed in operation in 2 days during August, 1972.

The Center Hill sources are also from an ongoing SHWRD experimental landfill project. Nineteen batch-type landfills containing 2 tons of municipal refuse, some with selected materials ($CaCO_3$, H_2O, industrial sludges) added, were constructed in November 1974 and April, 1975.

Small-scale experimental batch-type landfills were specially constructed (Figure 1) at the University of Cincinnati to investigate microbial population changes from the initial time of disposal. The contents of these landfills (Table 2) are combinations of or solely sewage sludge, municipal refuse, and shredded hospital ward wastes.

Leachate samples from all the test sites, except the University of Cincinnati

TABLE 1. CHARACTERIZATION OF COMMERCIAL AND EXPERIMENTAL LANDFILLS AND LABORATORY LYSIMETERS USED IN THIS STUDY

Type of Landfill or Lysimeter	Contained In	Dimensions	Solid Waste Added	Amount of Solid Waste	Date Prepared	Location of Site
Phase I						
Commercial Full-Scale Operational Landfill	Soil	700 Acres	Municipal, Hospital, Commercial	Millions of Tons	1954	[a]
Experimental Large-Scale, Batch-Type, Field Sanitary Landfill	Soil	40'x140' x8.5' deep	Municipal	435 Tons	June, 1971	Walton, Kentucky
Experimental Field Large-Scale, Batch-Type Lysimeters	Steel Cylinders in Soil	12'x6'	Municipal	2 Tons	August, 1972	Walton, Kentucky
Experimental Field Large-Scale, Batch-Type Lysimeters	Steel Cylinders in Soil	12'x6'	Municipal, Sewage Sludge, Industrial	2 Tons	November, 1974 and April, 1975	Center Hill, Cincinnati, Ohio
Phase II						
Experimental, Laboratory, Small-Scale Lysimeters	Steel Drums at 20°C in the Laboratory	2'x2'	Municipal, Hospital, Sewage Sludge	150 Lbs.	August, 1978	University of Cincinnati

[a] Name withheld since commercial firm involved.

FIGURE 1

Schematic of Experimental Laboratory Lysimeter

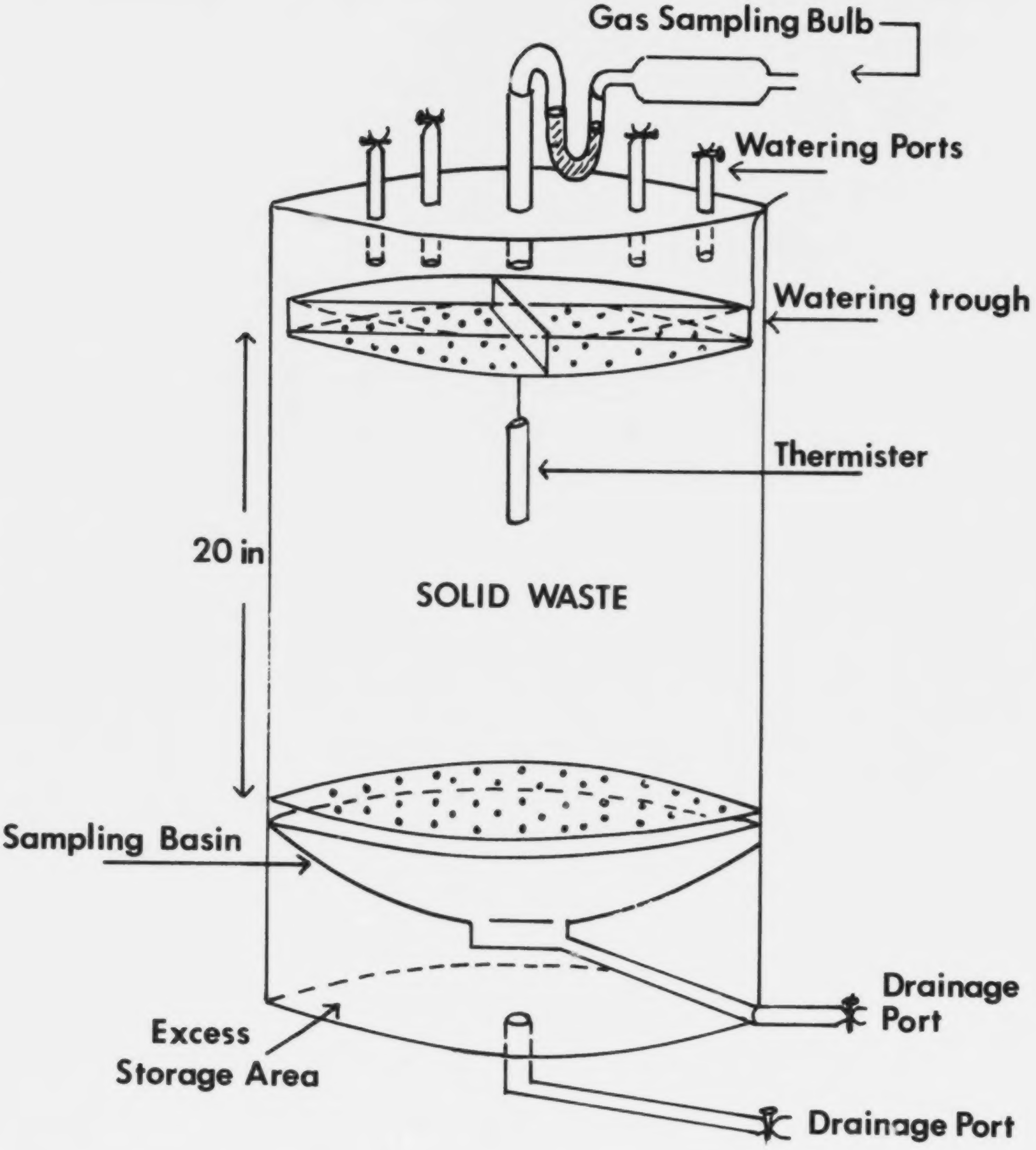
Gas Sampling Bulb
Watering Ports
Watering trough
Thermister
20 in
SOLID WASTE
Sampling Basin
Drainage Port
Excess Storage Area
Drainage Port

TABLE 2. CONTENTS OF EXPERIMENTAL LABORATORY - SCALE, BATCH-TYPE SANITARY LANDFILL TEST CELLS (LYSIMETERS) AT THE UNIVERSITY OF CINCINNATI

Lysimeter[a]	Contents
A	Sewage Sludge
B	Municipal Solid Waste and Sewage Sludge
C	Municipal Solid Waste
D	Municipal Solid Waste
E	Municipal Solid Waste and Hospital Waste
F	Hospital Waste

[a] Constructed in August, 1978, and placed at 20°C.

lysimeters, were each aseptically collected four times a year. The samples were refrigerated within one hour after collection, and were analyzed within 24 hours. The leachates in the University of Cincinnati study were collected on a frequent basis, often weekly, and were examined within 24 hours. All leachates were diluted and plated using the bacteriological smear technique both aerobically and anaerobically on blood agar plates, and on other differential agar plates. Cooked meat medium and selenite F were used also. Coliform and fecal streptococcal MPN tests were included. Chemical analyses were made to characterize the leachates.

The microorganisms obtained for identification came from the aerobic and anaerobic blood plates, selenite F, cooked meat medium and azide dextrose medium. The procedures for the identification of microorganisms present in leachate came from Microbiological Methods for Monitoring the Environment (1978), Standard Methods for Examination of Water and Wastewater (1976), and current clinical manuals. API strips for the enterobacteraceae and the clostridia were also used.

EXPERIMENTAL RESULTS

The criteria used to determine if an isolated organism was pathogenic was if it appeared on the U. S. Public Health Service's Classification of Etiologic Agents on the Basis of Hazard (1976). This classification is based on how infectious the microorganisms are for humans and animals, as shown in Table 3. Bacteria from Class 2, moderately infectious organisms, which could be expected to be found in municipal refuse are listed in Table 4. Those organisms which could be found in human or animal feces were listed in this table, along with respiratory organisms which might find their way to refuse.

Certain species of streptococci occur only in man, and/or animals, fowls, and/or vegetation. Therefore, such information is useful for source identifications. Table 5 shows the human, animal, and vegetable origins of streptococcal groups.

Microbes Isolated and Identified

The microorganisms isolated from the landfills are shown in Tables 6 and 7. They are grouped by the landfills from which they were obtained, and are listed in alphabetical order. The four most frequent species isolated, listed in the order most frequent, are: species of the genera Bacillus, Corynebacterium, and Streptococcus, and yeast. The yeast isolates were not differentiated into genera. It was usual to find only 3 or 4 genera in a leachate sample. Note on the tables that some of the microorganisms are marked with an asterisk. These appear in the Public Health Service publication Classification of Etiologic Agents on the Basis of Hazard (1976) as Class 2 bacterial agents. These are Listeria monocytogenes, Moraxella, and Acinetobacter. Allescheria boydii (see Table 7), while not on the CDC list, is a pathogen which causes madura foot abscesses. In general, all these microorganisms are not particularly invasive, but they are pathogenic.

These landfills were over 4 years old, yet many types of microorganisms were present. The spore-bearing microorganisms were the Bacillus sp., the Clostridia sp., and the fungi. These are known to endure in the presence of high temperatures, disinfectants, etc. The Listeria monocytogenes is known to be a particularly persistent microorganism in refrigerated cultures, and are known to grow at low temperatures.

Assays on leachate from the older landfills (Phase I) were begun in November, 1977, and indicated that total coliform, and consequently fecal coliform counts, were less than 30/100 ml. Previous investigations of the coliform content of the Batch-Type Field Sanitary Landfill leachates by others indicated they were not detectable 30 weeks after construction of the landfill. In Phase II of this study, the small scale landfills at the University of Cincinnati showed total coliform titers of 10^9 from initial leachate samples taken from the municipal solid waste lysimeters. The titer was 10^3 three weeks later, 10^2 after four weeks, and less than 20 after six weeks.

TABLE 3. THE BASIS FOR CLASSIFICATION[a] OF HAZARDOUS MICROBIAL AGENTS

CLASS	DEFINITION
1	Agents of no or minimal hazard to human or animal health.
2	Agents of ordinary potential hazard such as staphylococci, that can cause disease when the agent penetrates the skin.
3	Agents of special hazard such as human tuberculosis which requires special conditions in order to contain them.
4	Agents of extreme hazard to personnel or may cause serious epidemic disease. Example: smallpox virus.
5	Foreign animal pathogens which cannot be permitted in the United States. Example: foot and mouth disease virus.

[a] From Classification of Etiologic Agents on the Basis of Hazard, 1976.

TABLE 4. REPRESENTATIVE PATHOGENIC[a] MICROORGANISMS WHICH MAY BE ISOLATED FROM MUNICIPAL LANDFILLS

Microorganisms	Natural Habitat in		Highly Infectious	Disease Produced
	Nature	Man		
Clostridium sp.	+	+	-	Food poisoning, diarrhea, gas gangrene
Erysipelothrix insidiosa	+	+	-	Skin infection, arthritis, septicemia
Klebsiella pneumoniae	+	+	+	Pneumonia
Listeria sp.	+	-	-	Abscesses
Mima polymorpha (*Actinetobacter sp.*)	-	+	-	Skin, eye infection, pneumonia
Moraxella sp.	-	+	-	Urinary tract infection
Pasteurella sp.	-	+	-	Gastrointestinal disturbance
Salmonella sp.	+	+	+	Food poisoning, septicemia
Shigella sp.	-	+	+	Intestinal infection
Staphylococcus aureus	-	+	-	Wound infection, food poisoning
Streptococcus pyogenes	-	+	+	Acute infection

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[a] As defined by the CDC Classification of Etiologic Agents.

TABLE 5. THE HUMAN, ANIMAL, & VEGETABLE ORIGINS OF STREPTOCOCCAL GROUPS

Group[a]	Species	Origin
A	*S. pyogenes*	man
B	*S. agalactiae*	cattle, man
Not Groups A, B, or D (i.e. Groups C, G, F)	*S. dysagalactiae*	man, animal
	S. equi	
	S. equisimilis	
	S. zooepidemicus	
	S. anginosus	
D, not enterococcus	*S. bovis*	animal
	S. equinus	horse
Enterococcus (Group D)	*S. faecalis*	man, animal
	S. faecalis subsp. *liquefaciens*	vegetarian, insects
	S. faecalis subsp. *zymogenes*	man, animal
	S. faecium	man, animal
	S. durans	man, animal
Viridans	*S. salivarius*	man
	S. mitis	man
Q	*S. avium*	fowl

[a] Lancefield Groups A through Q are determined by the precipitan test.

Viridans is determined by henolysis on blood agar plates.

TABLE 6. BACTERIA ISOLATED FROM COMMERCIAL AND EXPERIMENTAL LANDFILL LEACHATES

COMMERCIAL SITE	*Achromobacter* sp.
	Anaerobic streptococcus
	Bacillus sp.
	Clostridium fallax
	Clostridium sp.
	Corynebacterium sp.
	pseudomonas - like
	Streptococcus sp.
	α streptococcus
BOONE COUNTY FIELD SITE	*Aeromonas* sp.
	Bacillus sp.
	CDC Group Ve-1
	Corynebacterium sp.
	**Listeria monocytogenes*
	Micrococcus sp.
	**Moraxella* sp.
	Neisseria sp
	Serratia marcescens
	Streptococcus sp.
	α streptococcus
CENTER HILL FACILITY	**Acinetobacter* sp.
	Bacillus sp.
	Clostridium sp.
	Corynebacterium sp.
	Enterobacter cloacae
	**Listeria monocytogenes*
	pseudomonas - like
	Staphylococcus albus
	Streptococcus sp.

*CDC Agents of ordinary potential hazard.

TABLE 7. FUNGI ISOLATED FROM COMMERCIAL AND EXPERIMENTAL LANDFILL LEACHATES

COMMERCIAL SITE	Yeast
BOONE COUNTY	*Allescheria boydii* *Cephalosporium* sp. Yeast
CENTER HILL FACILITY	*Fusarium* sp. *Penicillium* sp. *Sepedonium* sp. Yeast

Obviously, the total coliform (and the fecal coliform) MPN tests were not desirable as indicators of pathogenic presence in landfill leachates because die-off was more rapid than the pathogens which were later isolated.

Unlike the coliforms, fecal streptococci were found in most all leachates sampled. The pathogens listed in Table 6 were frequently found in the presence of fecal streptococci; therefore, the fecal streptococci assay is a better indicator of the presence of pathogens than fecal coliform. Results of streptococcal assays of the Center Hill leachates (Figure 2) indicated they were not always present in all leachates, regardless of whether the leachates derived from municipal refuse or mixtures of refuse and selected industrial sludges. This indicates the possibility of pathogen presence in the absence of fecal streptococci, dependent on the ability of pathogens to withstand the environment which depressed the fecal streptococci.

Assay Interferences

The absence of fecal streptococci in some of Center Hill assays was surprising and warranted further investigation (Figure 2). While some of the experimental landfills contained heavy metal bearing wastes, the negative results of the fecal streptococci assays occurred in the municipal refuse leachates as well as from the refuse-industrial sludge wastes.

A Gram stain of all of the positive ethyl violet azide tubes showed that some of the positive tests did indeed contain streptococci, while a large number of others showed Gram positive rods instead of cocci. Litsky et al. (1953) found that upon the examination of sewage and river water that Gram positive rods growing in the presumptive test azide dextrose broth tubes could also grow out in confirmatory ethyl violet azide broth and produce a positive reaction, i.e., a purple button in the bottom of the tube.

Leachates from the municipal solid waste test cells at Center Hill that were positive for fecal streptococci were used as an undiluted sample in the test procedure, and were found to inhibit streptococcal growth. After the leachate was diluted to 10^{-1} or 10^{-2}, the streptococci did grow out in the azide dextrose and in the ethyl violet azide broth to produce a positive test for streptococci. In addition, the large number of very low streptococci titers, especially in the Center Hill test cells containing municipal solid waste, i.e., Test Cells 3, 11, 15, and 16, suggested that the streptococcal levels were becoming low and that the streptococci enter the leachate in an irregular fashion, and thus do not always appear in the leachate. It should be mentioned also that the numbers of Gram positive sporulating Bacillus sp., recovered on plate counting medium far exceeded those of the other microorganisms present. Perhaps the large number of these rods interfere with the growth of the fecal streptococci.

Some preliminary work was done to determine if leachate itself could interfere with the isolation of pathogens, since so few were present and they appeared in such an irregular fashion. The study consisted of adding washed cells of Klebsiella pneumoniae, Salmonella typhimurium, Pseudomonas aeruginosa, and Streptococcus faecalis to both phosphate buffer and leachate held at room temperature. Recovery of the microbes commenced immediately after the dilutions were prepared. As seen in Figure 3, the cultures in the leachate decreased at least 2 logs. The recovery counts in leachate were 10^5/ml while those placed in buffer had a count of 10^7 or more. This indicated possible interference of the leachate with the recovery procedure, and the possible inhibition of pathogen growth. Other investigators (Glotzbecker, 1974) in our laboratory showed that only 1% Escherichia coli and 10% Streptococcus faecalis survived 2 hours in Boone County leachate at 10°C. Possible explanations for this apparent toxicity could include (1) metabolic inhibition due to drastic changes in the chemical environment from the growth medium to the leachate; (2) toxic levels of specific chemical constituents of the leachate; and (3) biological competition for nutrients by other, more

FIGURE 2

Fecal Streptococcal Presence in Center Hill Experimental Field Lysimeter

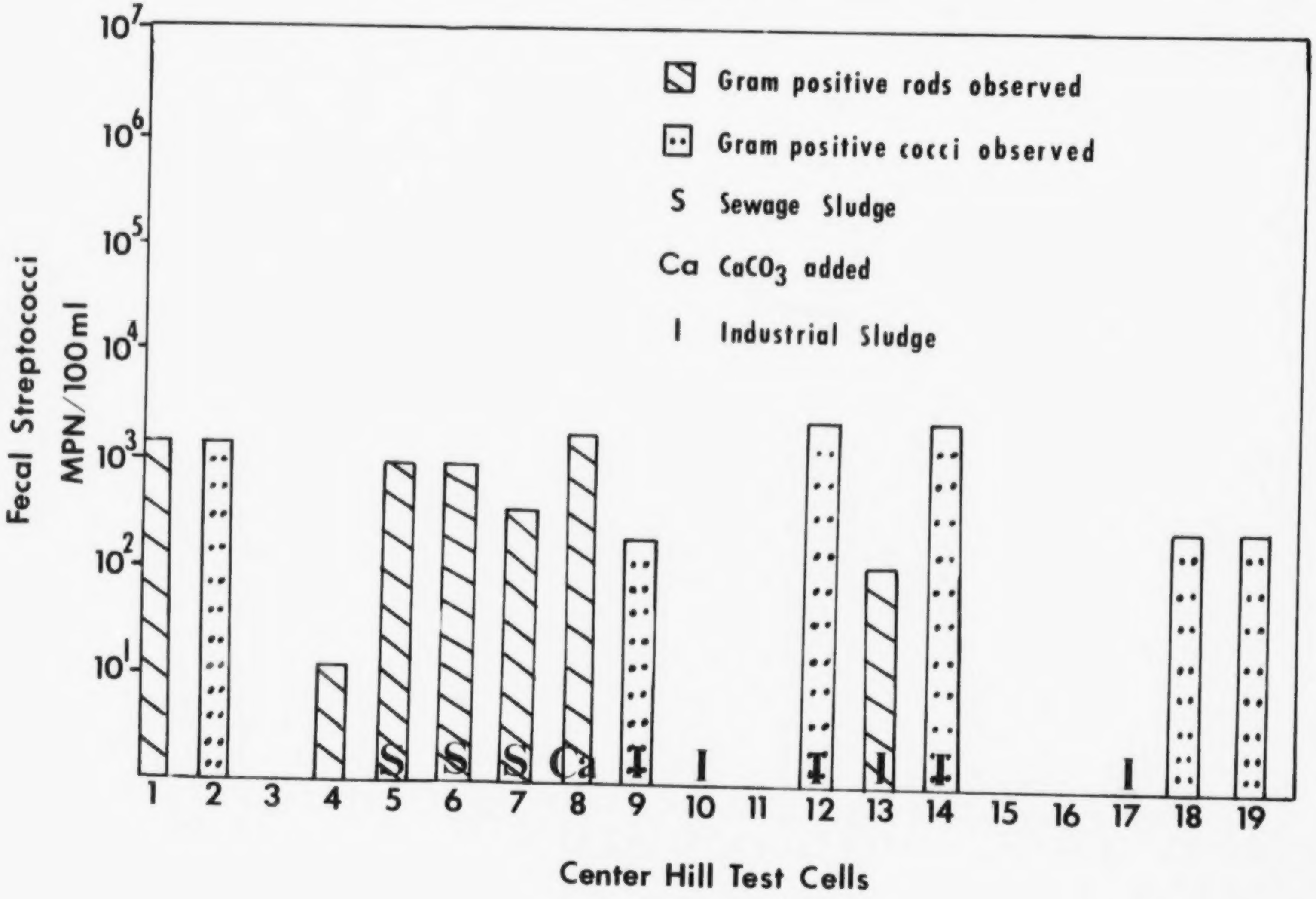
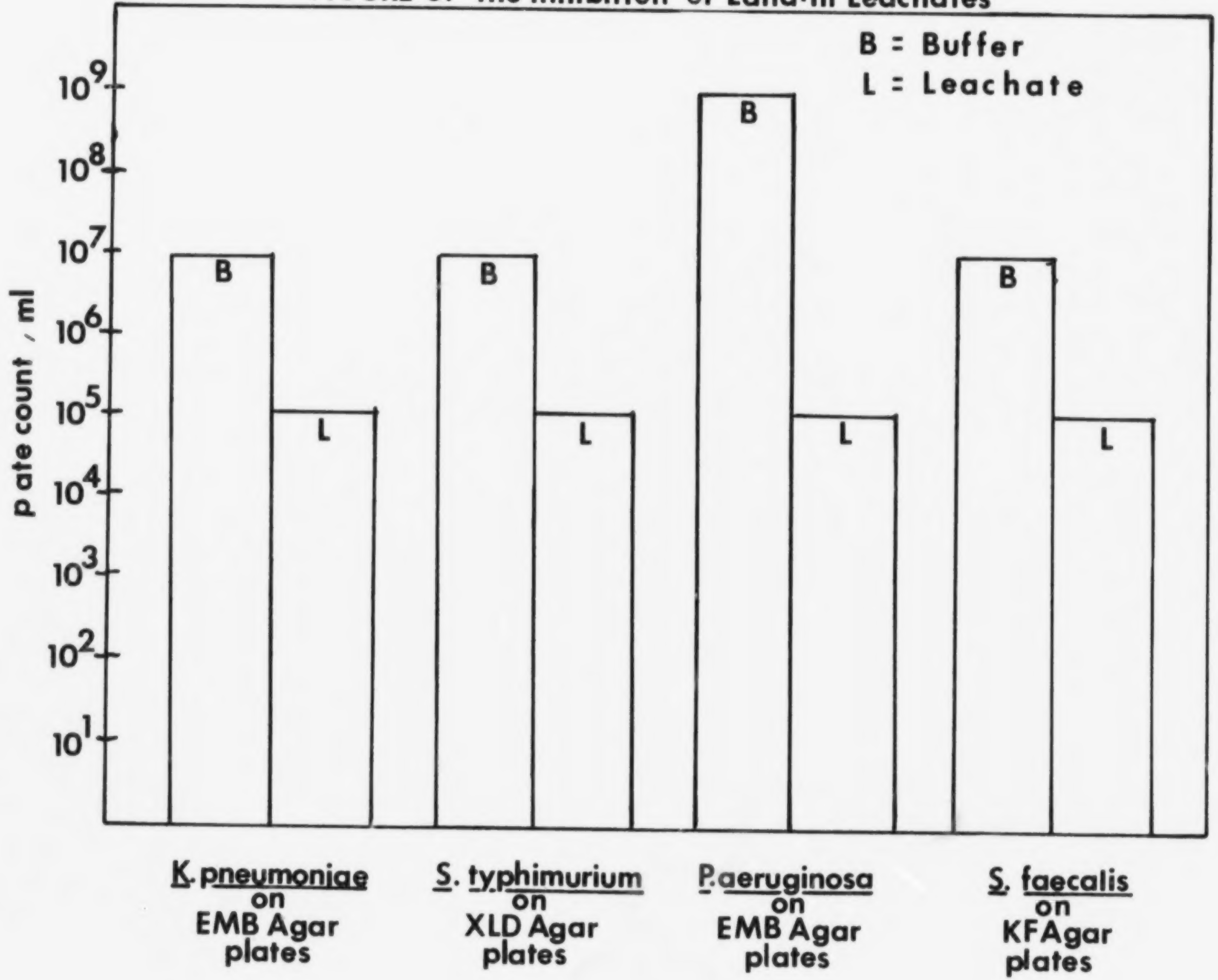


FIGURE 3. The Inhibition of Landfill Leachates



numerous organisms. Suppression of growth or even reduction in the numbers of organisms detected does not necessarily indicate death, but merely an inability to ascertain active growth or metabolism. A change in the environment, such as achieved by dilution in a less antagonistic liquid (buffer solution, clean water, etc.), can lead to more diverse identification and enumeration. Alternatively, long term exposure to the antagonistic conditions of leachate may effectively reduce the number of pathogens and indicators released from a landfill.

Phase II Assays

The specially constructed experimental landfills were constructed in August, 1978 and kept in a 20°C environmental room.* The landfills received 20 inches (508 mm)/yr. of net infiltration (Figure 4). The wastes were brought to field capacity over a 6 week period. Leachates were collected for analysis on a frequent basis, often weekly.

Bacterial counts of the solid wastes used in the construction of the lysimeters are reported in Table 8. The numbers of total viable aerobic bacteria, total and fecal coliforms, and fecal streptococci recovered were very similar, regardless of waste type; only fecal streptococci seemed to show a significant difference. The species of gram negative bacteria recovered from the sewage sludge, and the municipal and hospital wastes used in the construction of the lysimeters are listed in Table 9. Note the presence of the pathogens. CDC Class II Klebsiella sp., were found in all three wastes; Pasteurella hemolyticum was found only in sewage sludge; Mima sp., was found in both sewage sludge and refuse; the other pathogens isolated were found in both hospital waste and refuse. Thus, pathogens may be expected to be found in landfilled wastes.

The isolation and speciation of fecal streptococci are shown in Table 10, but this study is not completely meaningful at this time because of the small number of colonies which have been identified. This work is continuing and should allow for more complete characterization of the wastes placed in the landfills.

Numbers of bacteria quantified over the initial 17 week period of waste decomposition are provided in Tables 11, 12, and 13. The aerobic plate counts showed a gradual decline in the number of organisms over the 17 week period. Leachates from sewage sludge generally yielded slightly smaller numbers of bacteria than when mixed with municipal refuse. Leachates from hospital waste generally yielded slightly larger numbers than leachates from the hospital waste-municipal refuse mixture. The number of bacteria assayed with blood agar plates also showed a gradual decline over the 17 week period. A change in the method of analysis for total and fecal coliform (to that of Allen et al., 1976) resulted in slightly larger enumeration in weeks 13, 15, and 17, but the decline in the number of organisms over the 17 week period was very pronounced for the coliforms and less rapid for the fecal streptococci. The municipal refuse-sewage sludge mixture had larger numbers of indicator organisms than did the sewage sludge alone, but the two sources of leachate yielded approximately the same numbers by week 17. The somewhat erratic numbers of fecal streptococci indicated that they can thrive and multiply in some landfill environments.

The data summarized in Table 14 is incomplete, since the fecal streptococcal speciation studies are in progress. Fecal streptococci have been observed in leachates from all laboratory lysimeters in higher numbers than the fecal or total coliforms. We hope in the near future to accumulate more complete speciation data so that we may draw final conclusions as to the origins of the fecal material in the lysimeters. However, at this time eighty-five species of streptococci were isolated from these experimental lysimeters, and fifty-seven species or 67 percent were enterococci. This indicated the presence of fecal matter in the lysimeter contents, and thus indirectly the presence of bacterial pathogens. No pathogens have been isolated yet; assays are continuing. However, Table 15 indicated the presence in these leachates of diphtheroids of unknown etiology which have been found but not identified. Therefore, we will have to delay final conclusions as to the presence of pathogens until pathogenic identification

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*The temperature of the landfills stabilized at 20°C one week after construction.

FIGURE 4

Addition of Water and Production of Leachate in Laboratory Lysimeter C

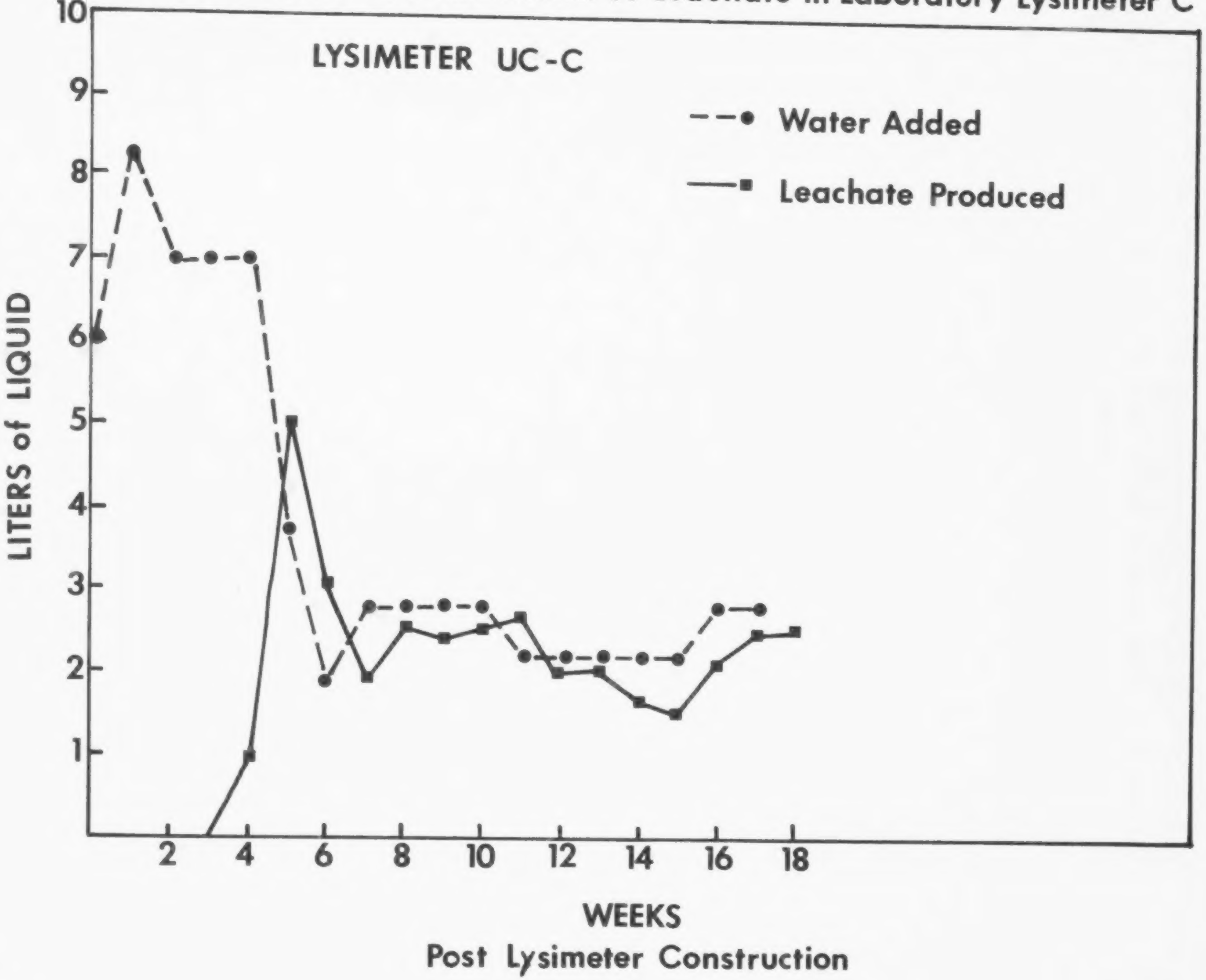

TABLE 8. BACTERIAL COUNTS OF THE SOLID WASTE USED IN CONSTRUCTION OF THE UNIVERSITY OF CINCINNATI LYSIMETERS

Solid Waste	Bacterial Counts			
	MPN/gram			
	Total Coliform	Fecal Coliform	Fecal Streptococcal	Bacterial Numbers/gram[a]
Sewage sludge	2.8×10^{11}	2.3×10^{10}	3.3×10^{9}	1.67×10^{9}
Hospital waste	9.0×10^{10}	9.0×10^{10}	8.6×10^{10}	3.8×10^{9}
Municipal waste	7.65×10^{10}	4.65×10^{10}	2.5×10^{11}	4.28×10^{10}

[a] Recovered on brain heart infusion agar plates.

TABLE 9. GRAM NEGATIVE BACTERIA ISOLATED FROM SOLID WASTE USED IN THE CONSTRUCTION OF THE UNIVERSITY OF CINCINNATI LYSIMETERS

Bacterial Genera and Species	Numbers Isolated From		
	Sewage Sludge	Municipal Waste	Hospital Waste
Oxidase Negative Lactose Fermenters			
Escherichia coli.	8	3	2
Enterobacter sp.	7	3	6
**Klebsiella* sp.	3	3	5
Citrobacter sp.	4	10	11
Oxidase Negative Non-lactose Fermenters			
Serratia sp.	4	b	4
Proteus sp.	3	10	13
Providance sp.	b	1	1
**Salmonella* sp.	b	3	1
Oxidase Positive Non-fermenters			
Aeromonas[c] sp.	b	2	3
Flavobacterium sp.	1	b	b
**Herellea* sp.	b	2	2
**Mima* sp.	2	1	b
**Moraxella* sp.	b	1	1
**Pasteurella hemolyticum*	1	b	b
Pseudomonas sp.	1	2	17

[a] Identified by the procedure of Wolf (1975).

[b] None detected.

[c] A fermenter.

*These bacteria appear on the CDC list classifying etiologic agents on the basis of hazard (Classification of Etiologic Agents on the Basis of Hazard, 1976).

TABLE 10. STREPTOCOCCAL GROUPS AND SPECIES ISOLATED FROM SOLID WASTE USED IN THE CONSTRUCTION OF THE UNIVERSITY OF CINCINNATI LYSIMETERS

Streptococcal Groups[a] and Species[a]	Numbers of Streptococcal Isolates From	
	Municipal Waste	Hospital Waste
Group A	[b]	[b]
Group B	1	[b]
Not Groups A, B, or D (i.e. Group C, G, and/or F)	2	1
Group D (enterococcus)		
S. faecalis subsp. *liquefaciens*	3	5
S. faecalis	1	[b]
S. faecium	1	[b]
Total	5	5
Group D (not enterococcus)	1	1
Viridans		
S. salivarius	2	1
Total Viridans Group	5	5

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[a] The streptococcal groups were identified by the procedure of Facklam (1974) and the species were identified by the procedure of Pavlova (1972).

[b] None detected.

TABLE 11. BACTERIAL NUMBERS IN LEACHATE FROM UNIVERSITY OF CINCINNATI LYSIMETERS

Leachate Sample Taken at Week[a]	Bacterial Numbers/ml[b] in Leachate From Lysimeter[c]					
	A	B	C	D	E	F
4	2.8×10^{4}	1.5×10^{7}	5.8×10^{5}	1.5×10^{7}	d	d
5	2.0×10^{6}	2.2×10^{6}	4.7×10^{7}	8.0×10^{7}	1.6×10^{7}	8.5×10^{6}
7	6.9×10^{5}	3.4×10^{6}	8.4×10^{6}	5.7×10^{6}	7.2×10^{6}	2.6×10^{7}
8	2.5×10^{6}	4.2×10^{6}	5.5×10^{6}	d	4.0×10^{6}	1.58×10^{7}
9	1.5×10^{6}	1.8×10^{6}	4.0×10^{6}	3.4×10^{5}	8.3×10^{5}	1.0×10^{5}
10	4.3×10^{5}	1.0×10^{6}	2.75×10^{5}	1.2×10^{6}	5.6×10^{5}	1.9×10^{6}
11	4.3×10^{5}	2.7×10^{6}	2.0×10^{5}	3.2×10^{5}	4.3×10^{5}	5.1×10^{5}
12	7.0×10^{5}	7.35×10^{5}	4.0×10^{5}	7.1×10^{5}	3.2×10^{5}	1.43×10^{6}
13	1.4×10^{6}	5.4×10^{5}	7.9×10^{5}	3.1×10^{6}	1.46×10^{5}	6.3×10^{6}
14	8.6×10^{4}	3.9×10^{5}	1.06×10^{6}	2.75×10^{6}	2.65×10^{4}	3.9×10^{5}
15	3.5×10^{5}	3.8×10^{5}	3.2×10^{5}	3.5×10^{6}	1.0×10^{5}	3.2×10^{6}
17	9.7×10^{5}	2.5×10^{5}	8.2×10^{5}	1.6×10^{5}	5.3×10^{4}	2.8×10^{6}

[a] Weeks after construction of lysimeters.

[b] Determined with plate count agar incubated at 35°C under aerobic conditions.

[c] Lysimeter contents:
A = sewage sludge
B = municipal solid waste plus sewage sludge
C and D = municipal solid waste
E = municipal solid waste plus hospital waste
F = hospital waste

[d] Not determined.

TABLE 12. NUMBERS OF AEROBIC AND ANAEROBIC BACTERIA IN LEACHATE FROM UNIVERSITY OF CINCINNATI LYSIMETERS

Leachate Sample Taken at Week[a]	Conditions of Incubation[b]	Bacterial Numbers/ml[c] in Leachate From Lysimeter[d] A	B	C	D	E	F
4	Aerobic	1.6×10^{4}	4.3×10^{5}	2.2×10^{5}	1.5×10^{7}	e	e
	Anaerobic	1.0×10^{3}	2.6×10^{8}	2.3×10^{7}	6.9×10^{7}	e	e
5	Aerobic	1.5×10^{5}	2.4×10^{6}	1.8×10^{7}	1.5×10^{7}	1.0×10^{7}	2.2×10^{7}
	Anaerobic	1.5×10^{5}	1.3×10^{7}	4.4×10^{7}	6.7×10^{7}	3.1×10^{7}	4.5×10^{7}
8	Anaerobic	8.1×10^{4}	2.1×10^{6}	3.0×10^{7}	2.8×10^{7}	1.3×10^{7}	2.0×10^{7}
9	Aerobic	7.5×10^{4}	1.0×10^{6}	7.9×10^{5}	1.2×10^{5}	1.4×10^{5}	1.0×10^{5}
	Anaerobic	6.6×10^{3}	3.1×10^{4}	1.6×10^{6}	3.4×10^{5}	8.8×10^{5}	1.3×10^{6}
10	Anaerobic	2.2×10^{4}	1.1×10^{5}	1.4×10^{5}	2.1×10^{5}	1.2×10^{6}	1.1×10^{5}
12	Anaerobic	5.0×10^{3}	1.3×10^{5}	3.2×10^{5}	5.0×10^{4}	7.4×10^{4}	1.5×10^{5}
13	Aerobic	1.9×10^{5}	1.2×10^{6}	6.8×10^{5}	2.7×10^{6}	7.6×10^{4}	1.2×10^{5}
	Anaerobic	6.3×10^{3}	1.1×10^{5}	8.0×10^{5}	5.6×10^{5}	3.1×10^{5}	2.1×10^{6}
14	Anaerobic	3.0×10^{3}	1.7×10^{5}	4.6×10^{5}	2.4×10^{5}	1.5×10^{5}	1.0×10^{6}
15	Anaerobic	4.0×10^{4}	6.0×10^{4}	1.2×10^{5}	1.36×10^{5}	6.0×10^{4}	4.2×10^{5}
17	Aerobic	3.74×10^{5}	6.3×10^{4}	7.9×10^{5}	1.76×10^{7}	3.0×10^{4}	6.2×10^{4}
	Anaerobic	3.76×10^{5}	e	9.4×10^{4}	3.1×10^{5}	3.0×10^{5}	1.56×10^{5}

[a] Weeks after construction of lysimeters.

[b] Incubation at 35°C for 48 hours. The GasPak provided anaerobic conditions.

[c] Determined with blood agar plates.

[d] Lysimeter contents: A = sewage sludge, B = municipal solid waste plus sewage sludge, C and D = municipal solid waste, E = municipal solid waste plus hospital waste, F = hospital waste.

[e] Not determined.

TABLE 13. COLIFORM AND STREPTOCOCCAL MPN TITERS[a] IN LEACHATES FROM UNIVERSITY OF CINCINNATI LYSIMETERS[b]

	Leachate MPN Counts:					
	A			B		
Weeks[c]	Total Coliform	Fecal Coliform	Fecal Streptococcal	Total Coliform	Fecal Coliform	Fecal Streptococcal
4	5	2	$7.9x10^{2}$	$1.4x10^{5}$	$9.4x10^{4}$	$4.9x10^{5}$
5	< 20	< 20	$9.2x10^{3}$	$1.3x10^{2}$	$5.0x10^{1}$	$9.2x10^{2}$
7	< 20	< 20	$9.2x10^{2}$	$1.7x10^{2}$	2	$9.2x10^{4}$
9	< 20	< 20	$2.4x10^{3}$	< 20	< 20	$1.6x10^{4}$
11	< 20	< 20	$3.5x10^{3}$	< 20	< 20	$3.5x10^{3}$
13	7[d]	< 20[d]	$5.4x10^{2}$	$8.0x10^{2}$[d]	< 20[d]	$9.2x10^{3}$
15	$2.2x10^{2}$[d]	$1.7x10^{1}$[d]	$3.5x10^{2}$	< 2[d]	< 2[d]	$2.3x10^{3}$
17	$1.3x10^{2}$[d]	< 2[d]	$1.7x10^{2}$	< 2[d]	< 2[d]	$1.7x10^{2}$

[a] Coliform and streptococcal MPN/100 ml as determined by Standard Methods (1976).

[b] Lysimeter contents: A = Sewage sludge
B = Municipal solid waste plus sewage sludge

[c] Weeks after Lysimeter construction.

[d] Procedure changed according to Allen et al (1976).

TABLE 14. STREPTOCOCCAL GROUP AND SPECIES ISOLATED FROM LEACHATES OBTAINED THROUGH THIRTEEN WEEKS AFTER CONSTRUCTION OF THE UNIVERSITY OF CINCINNATI LYSIMETERS

Streptococcal Groups[a] and Species[b]	Numbers of Streptococcal Isolated from Lysimeter[b]					
	A	B	C	D	E	F
Group A	c	c	c	c	c	c
Group B	c	c	c	c	c	c
Non-group A, B, or D	c	c	c	c	c	c
Group D (enterococcus)						
S. faecalis subsp. liquefaciens	c	c	c	c	4	5
S. faecalis	1	1	c	c	3	8
S. faecium	3	5	1	c	4	c
S. durans	c	7	3	c	1	4
Total	4	16	4	c	16	17
Group D (not enterococcus)	1	c	5	7	4	1
Viridans						
S. salivarius	6	2	c	c	1	1

[a] The streptococcal groups were identified by the procedure of Facklam (1974) while the species were identified by the procedure of Pavlova (1972).

[b] The contents of lysimeters are:
A = sewage sludge
B = municipal solid waste plus sewage sludge
C = municipal solid waste
D = municipal solid waste
E = municipal solid waste plus hospital waste
F = hospital waste

[c] None detected at this time.

TABLE 15. NUMBERS OF NON-SPORE FORMING; GRAM POSITIVE; ROD SHAPED BACTERIA ISOLATED FROM UNIVERSITY OF CINCINNATI LYSIMETER LEACHATES

Leachate Sample Taken at Week[b]	Bacterial Numbers/ml[c] in Leachate From					
	Lysimeter[d]					
	A	B	C	D	E	F
4	$7.5x10^{3}$	e	e	e	e	e
5	$2.2x10^{4}$	$2.45x10^{4}$	$9.7x10^{4}$	$9.2x10^{4}$	$2.6x10^{4}$	$1.1x10^{5}$
9	$2.8x10^{4}$	$6.5x10^{3}$	$4.7x10^{4}$	$1.3x10^{4}$	$5.1x10^{4}$	$1.7x10^{4}$
13	$1.0x10^{4}$	$1.59x10^{4}$	$7.8x10^{4}$	$4.5x10^{5}$	$1.6x10^{4}$	$5.9x10^{4}$
17	$1.27x10^{4}$	$5.8x10^{3}$	$7.87x10^{4}$	$3.0x10^{4}$	$2.26x10^{5}$	$7.0x10^{6}$

[a] Examples: *Corynebacterium* sp., *Listeria* sp., *Kurthia* sp., etc.

[b] Weeks after construction of lysimeters.

[c] Determined on tellurite blood agar plates incubated for 48 hours at 35°C.

[c] Lysimeter contents:
A = sewage sludge
B = municipal solid waste plus sewage sludge
C and D = municipal solid waste
E = municipal solid waste plus hospital waste
F = hospital waste

[e] Not determined.

more complete speciation data so that we may draw final conclusions as to the origins of the fecal material in the lysimeters. However, at this time eighty-five species of streptococci were isolated from these experimental lysimeters, and fifty-seven species or 67 percent were enterococci. This indicated the presence of fecal matter in the lysimeter contents, and thus indirectly the presence of bacterial pathogens. Nevertheless, pathogenic isolations have not yet been made. However, Table 15 indicates the presence in these leachates of diphtheroids of unknown etiology. Therefore, we will have to delay final conclusions as to the presence of pathogens until pathogenic identification studies are completed.

Fungal (i.e., yeasts and molds) numbers were also determined using Sabouraud agar plates incubated at room temperature (20 to 22°C), and counted after one week. Even after 23 weeks, fungal numbers were still high (see Table 16).

Comparisons were also made of fecal coliform and fecal streptococci titers to determine the origin of fecal pollution in both the solid waste used to construct the lysimeters and in the lysimeter leachates. The latter studies (in the leachates) are not completed at this time, but the results of the former can be seen in Table 17. Note that the ratios of the sewage sludge (74) are indicative of pollution derived from domestic wastes composed of man's body wastes, whereas both the hospital and municipal wastes, if one holds to the ratio figures given in the literature, are not suggestive of human fecal pollution. Blannon and Peterson (1974) pointed out, however, that the FC:FS ratios must be applied carefully since these correlations may be altered because of the various ecological factors existing in solid waste landfills. For example, they noted in leachates a rapid die-off of fecal coliforms and the survival of fecal streptococci. It was suggested that the use of a ratio relationship for leachates may be valid only during the initial leaching period. Feachem (1975), however, presented data that showed that the differential die-away of fecal coliforms and fecal streptococci can, in fact, strengthen the value of the FC:FS ratio as a means of distinguishing human from non-human pollution. Feachem concluded that initially high FC:FS ratio values (>4) which fall indicate human contamination, whereas initially low ratios (<0.7) which subsequently rise suggest a non-human fecal source. Thus, reliance on the FC:FS ratios is possible even with differential survival. However, Feachem based his observations on FC;FS ratios in the water environment. It would also be significant if his observations could be extended to the landfill environment where changes in FC:FS ratios also occur.

The chemical environment in which the microbes were found was delineated to evaluate how chemical composition might affect microbial numbers and types. An example of the results showing the parameters evaluated is presented in Figure 5 for lysimeter E. Declining trends in the chemical parameters studied are apparent, but it is too soon to conclude what effect the chemical content of the leachates had on the microbial populations of the lysimeters.

CONCLUSIONS

Based on the above information, it can be concluded that:

1. A limited number of bacterial pathogens have been found in leachates from commercial and experimental landfills, and environmental lysimeters.
2. Total and fecal coliforms may not be acceptable indicators of fecal pollution after the first month or two of the landfill, since pathogens have been recovered from leachates with low or no total or fecal coliform numbers. Newer procedures for coliform recovery may indicate the presence of larger numbers.
3. The fecal streptococcal test still remains an attractive index of fecal (and pathogenic) pollution of leachates.
4. Leachate toxicity interferes with the isolation of microbes.

TABLE 16. FUNGAL NUMBERS ISOLATED FROM UNIVERSITY OF CINCINNATI LYSIMETER LEACHATES

Leachate Sample Taken at Week[a]	Fungal Numbers/ml[b] in Leachate From Lysimeter					
	A	B	C	D	E	F
5	2.0×10^4	5.0×10^4	1.5×10^4	3.0×10^5	< 10	3.0×10^4
13	1×10^3	1.3×10^5	c	4.0×10^5	4.6×10^4	7.0×10^4
15	5×10^3	1.6×10^4	c	1.2×10^4	2.0×10^4	1.8×10^4
23	1.4×10^4	3.2×10^4	1.5×10^4	1.13×10^5	1.1×10^5	1.0×10^5

[a] Weeks after construction of lysimeters.

[b] Determined on Sabouraud agar plates incubated at room temperature (20 to 22°C).

[c] Not determined.

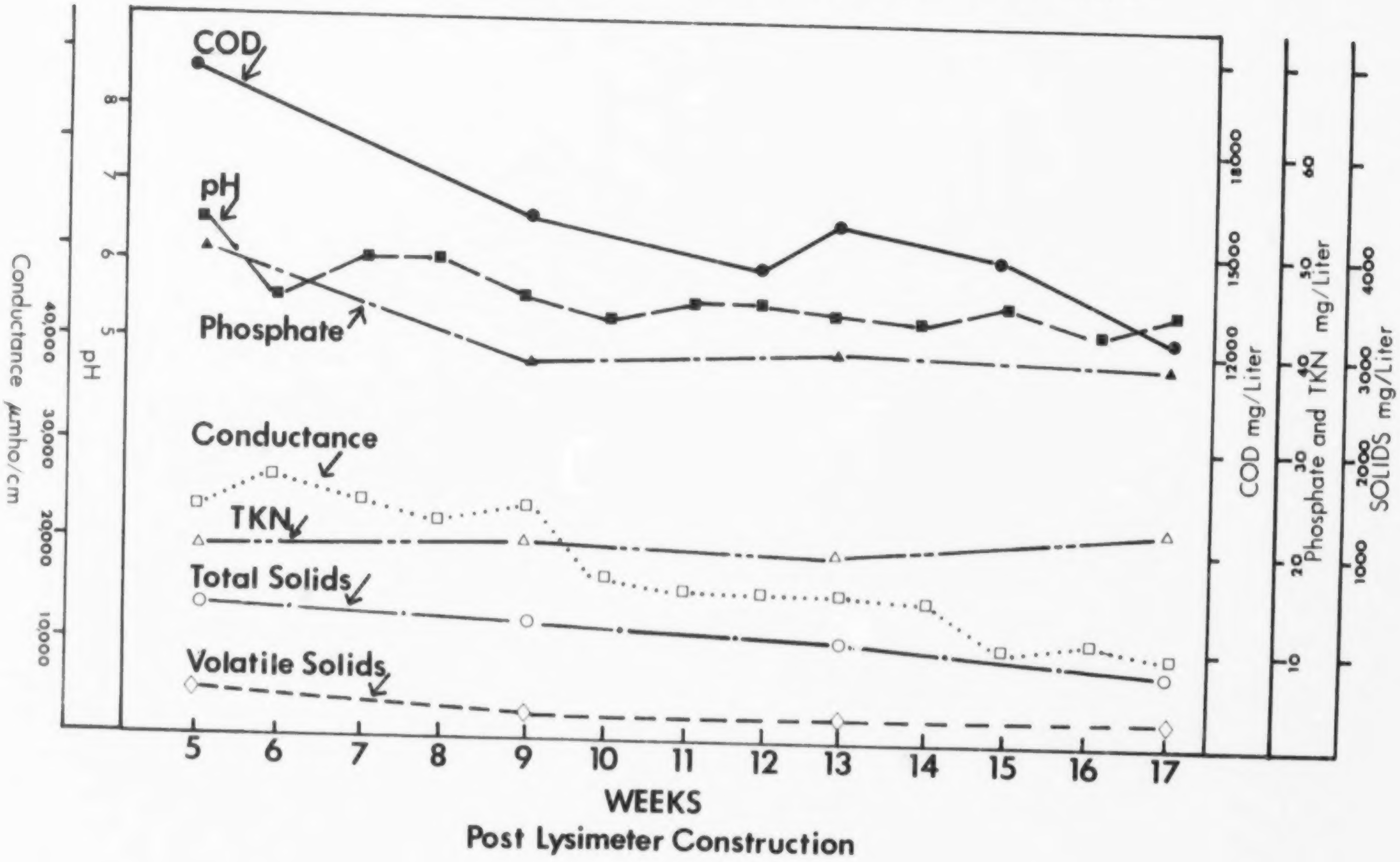
TABLE 17. COMPARISON OF FECAL COLIFORM AND FECAL STREPTOCOCCI TITERS TO DETERMINE THE ORIGIN OF FECAL POLLUTION OF SOLID WASTE IN UNIVERSITY OF CINCINNATI LYSIMETERS

Type of Solid Waste	Fecal Coliform/Fecal Streptococci Ratios[a]
Sewage sludge	$\frac{2.3\text{x}10^{10}}{3.3\text{x}10^{9}} = 6.96$
Hospital waste	$\frac{9.0\text{x}10^{10}}{1.1\text{x}10^{11}} = .82$
Municipal waste	$\frac{4.65\text{x}10^{10}}{2.5\text{x}10^{11}} = .19$

[a] Ratios of MPN values as described by Geldreich et al. (1964).

FIGURE 5

Chemical Analysis of Leachates from Laboratory Lysimeter E



by the standard plate count and blood agar plates generally decreased as disposal time increased. Fungal counts, however, remained relatively constant.

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GAS AND LEACHATE: SUMMARY

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ABSTRACT

Gas and leachate present a principal challenge to design and operation of sanitary landfills. Our knowledge of the behavior and characteristics of these contaminants is continually expanding and more effective ways to apply this knowledge are being found. Considerable variation exists in the physical processes affecting the fate of contaminants in different circumstances thus contributing to the difficulty in designing safeguards to the environment. For example, the harmful effects of leachates may be attenuated by some soils, but to rely upon soil properties as a sole design measure may be risky.

The variations in the properties of leachate and gas and the physical conditions under which they are generated and exist also complicates the design of collection and treatment devices that would lessen the harmful effects. Without knowing the precise parameters to design for, presents a risk of either over-designing or under-designing. In the first case excessive costs may be involved; in the latter, costs may be expended only to have the contaminants harm the environment anyway. This issue will become even more important as the Resource Conservation and Recovery Act provisions are promulgated and enforced for sanitary landfills.

Development of other answers and more precise application of the knowledge as it emerges must therefore continue. This paper discusses some of the past and current work and consolidates some findings of EPA to date. Even now the work is continuing and further breakthrough in the control and treatment of gas and leachate associated with sanitary landfills can be expected.

LANDFILL LEACHATE

Leachate is generated when water enters the landfill, percolates through it, and picks up soluble materials, some of them soluble products of biological and chemical reactions. Water can enter a fill by such means as precipitation or by drainage of floodwaters, springs, or groundwater into the fill. Therefore, sites are selected and sanitary landfills should be designed and constructed to avoid the intrusion of water.

Volume and Composition of Leachate

The volume of leachate produced depends on many factors, but is generally determined by the extent to which surface water infiltrates or groundwater is intercepted.

The composition of leachate varies widely and also depends on many factors such as the composition and age of the solid waste, temperature, and the amount of available oxygen. Commonly, there is little control over the quantity and characteristics of the input solid waste, so it is not surprising that output characteristics are so highly variable. Table 1 shows the range of compositions of leachates from sanitary landfills.

Examination of Table 1 leads to the following observations. Leachate is generally high in biologically oxidizable organics and total solids. Concentrations of heavy metals are generally less than 1 milligram per liter, but zinc occurs often in concentrations of 100 milligram per liter or more. Iron concentrations of 1000

TABLE 1. COMPOSITION OF TYPICAL LEACHATES FROM SANITARY LANDFILLS[1]

(mg/liter except pH value)

Constituent	Concentration range
Iron	200- 1,700
Zinc	1- 135
Phosphate	5- 130
Sulfate	25- 500
Chloride	100- 2,400
Sodium	100- 3,800
Nitrogen	20- 500
Hardness (as $CaCO_3$)	200- 5,250
COD	100-51,000
Total residue	1,000-45,000
Nickel	0.01-0.8
Copper	0.10-9.0
pH	4.00-8.5

milligrams per liter are common. Leachate derived from recently deposited solid waste is generally acidic, whereas the pH level of more stabilized solid wastes generally is near 7.

The biological properties of leachate have also been investigated, the studies focusing primarily on the density of total plate count bacteria, total coliforms, fecal coliforms, and fecal streptococci.[2-5] The data indicate a significant bacterial population associated with sanitary landfill leachate and also show a drastic reduction in the density of the bacteria with time of operation or with time of leaching. This decrease in density may result from adverse environmental conditions prevailing in the landfill.[6]

Migration of Leachate

From a technological point of view, the most difficult problem associated with design of a sanitary landfill is to predict the migration of contaminants in the soil. Computer simulation models have been developed to predict the movement of leachate through a landfill.[7] The fate of contaminants carried in solution and suspension is affected by physical processes, such as filtration, dispersion, and mixing, and by chemical processes in the soil, such as ion exchange, adsorption, and oxidation and reduction.

Mechanical filtration by soil is an effective process. Most suspended matter and a substantial number of microorganisms are removed from leachate after a few meters of travel through a fairly coarse, sandy soil. Although dispersion can reduce the concentration of pollutants in solution, it cannot be relied upon as a means of contaminant removal. Because of their large cation exchange capacity (CEC), and surface area, the clay minerals, e.g. montmorillonite, illite, kaolinite, chlorite, and vermiculite, are the most important fractions of soil in attenuating contaminants carried in leachate. Another important fraction is the hydrous iron oxide coatings on soil properties; this fraction is particularly effective in retaining metallic pollutants. Recent investigation[8] has led to the conclusion that passage of leachate through a calcium-saturated clay material will result in high attenuation of the heavy metals; moderate attenuation of potassium, ammonium, magnesium, and silicon; and relatively low attenuation of chlorine, sodium, and water-soluble organic compounds.

In addition to the clay and iron oxide related attenuation of pollutants in leachate, physical adsorption on surfaces is an important removal mechanism, particularly as regards bacteria and viruses. Again, the clay minerals are the most important fraction of soil.

This brief discussion of the attenuation process should indicate the complexity of the system and the difficulty of proceeding from a basic understanding of the system to a quantitative prediction of attenuation.

Environmental Effects of Leachate

Assessment of the environmental effects of leachate on surface and groundwaters indicates that pollution can occur unless adequate controls are provided during site selection and during the design, operation, surveillance, and long-term maintenance of a sanitary landfill. Detrimental effects of leachate on the environment include contamination of water supplies, fish kills in nearby streams, and loss of livestock.

In numerous reported instances, leachate has contaminated the surrounding soil and polluted an underlying groundwater aquifer or nearby surface water. The following are recent examples. Private wells located 300 meters downstream from the Llangollon landfill (completed and closed in 1968) in New Castle County, Delaware, were heavily polluted and subsequently abandoned. The municipal water supply from Geneseo, Illinois, was polluted by leachate from a garbage dump located about 500 meters north of the well field. Private wells located about 200 meters from a landfill in Kane County, Illinois, showed bacterial and chemical pollution.

Control and Treatment of Leachate

In many areas of the country, significant amounts of leachate are generated and few attenuating mechanisms are present. Systems have been developed for collecting and treating leachate in these less-than-ideal situations. Most collection systems utilize some type of impervious or highly impermeable liner.

Installation of an impermeable liner in a sanitary landfill to prevent leachate from polluting ground and surface waters is relatively recent development. Although laboratory studies have simulated field conditions in a sanitary landfill, little is known of the long-term effects of exposing liners to the many physical, chemical, and biological processes taking place within the landfill. The materials used as liners include compacted natural clay soils, compacted natural soils with bentonite added, asphaltic materials, and artificial membranes.

In one study, polymeric membranes and admix materials were exposed to sanitary landfill leachate, as shown in Table 2.[9]

After 1 year of exposure to leachate, most of the liner materials that were evaluated showed relatively minor deterioration. In no case was there a significant increase in water permeability. The polymeric membranes swelled and softened to various degrees, but did not become permeable. The admixed materials, such as soil, asphalt, and asphaltic concretes, showed significant loss of compressive strength but no increase in permeability. These results indicate that projections of the lifetime of the various liners when in contact with leachate must be based on much longer exposures.

Once leachate has been collected, it must be treated before it can be released. Treatment of leachate can be realized by:

- Recirculating the leachate back

TABLE 2. LINER MATERIALS SELECTED FOR LEACHATE EXPOSURE TESTS[9]

Material	Thickness, millimeters
Polymeric liner membranes:	
Polyethylene (PE)	0.10
Polyvinyl chloride (PVC)	0.20
Butyl rubber	0.63
Chlorosulfonated polyethylene, with fabric reinforcement	0.34
Ethylene propylene rubber (EPDM)	0.51
Chlorinated polyethylene (CPE)	0.32
Admix materials:	
Paving asphalt concrete	55.88
Hydraulic asphalt concrete	60.96
Soil cement	114.30
Soil asphalt	101.60
Bituminous seal	7.62
Emulsion asphalt or fabric	7.62

into the landfill to enhance anaerobic biological removal of the pollutants.

- Treating it with other wastes at a municipal wastewater treatment plant.
- Constructing a separate treatment unit at the landfill site.

When leachate is recirculated, it is applied to the top of the fill by surface irrigation. As a result, the moisture content of the solid waste is increased and presumably reaches an optimum level for anaerobic biological stabilization. In this treatment, the leachate is stabilized at a rate equal to that of the solid waste itself, with the solid waste functioning as an anaerobic trickling filter.

We have very little information on the combined treatment of leachate with domestic wastewater. One report[10] states that leachate with a COD of 10,000 milligrams per liter can be added to domestic wastewater in an extended aeration activated sludge unit at a level of at least 5 percent by volume without seriously impairing the effluent quality.

Construction of a separate on-site treatment unit is the most expensive method of treating leachate. Leachate is amenable to biological as well as physical-chemical treatment processes. Following a recent study at the University of Illinois, Chian and DeWalle[11] concluded that biological treatment methods, such as use of anaerobic filters, aerated lagoon, and combined treatment, are most applicable for treating leachate from recently constructed landfills, whereas physical-chemical treatment methods, such as activated carbon adsorption, chemical precipitation, chemical oxidation, and reverse osmosis, are best applied to treatment of leachate from more stabilized solid waste.

Because the magnitude of any leachate problem depends on many variables, the effects of leachate collection and treatment on the cost of sanitary landfill operations are highly site-specific.

Chian and DeWalle[12] found that aerated lagoons provide the least expensive method of treating leachates having comparatively low BOD_5 values, (e.g. 5000 mg/liter) and relatively high flowrates (e.g., 1.26 liters per second). As the BOD_5 level of the leachate increases at the same high flowrate, the cost of treatment with anaerobic filters becomes increasingly attractive, and at a BOD_5 value of 25,000 milligrams per liter it equals that of the aerated lagoon process when credit is deducted for the methane gas produced. Treatment of leachate in aerated lagoons and by physical-chemical treatment processes is most effective, although such treatment would cost more than combined treatment of leachate and domestic wastewater with activated sludge at high leachate flowrates and low BOD_5 levels. The cost estimates are summarized in Table 3.

LANDFILL GAS

Gas is a natural decomposition product resulting from the microbial degradation of solid waste. Although such degradation produces a variety of gases, methane and carbon dioxide are the major gaseous products of landfill decomposition. Migration of gas produced in a landfill may lead to a variety of effects on the environment, ranging from persistant problems with repugnant odors to the accumulation and explosion of high concentrations of methane. Because of the potential problems, both the production and migration of gas are important considerations in selecting a site and operating a landfill.

Volume and Composition of Landfill Gas

Gases generated from the stabilization of landfilled solid waste are the result of microbial degradation of the organic matter present in the waste. Although aerobic processes occur initially, facultative and anaerobic processes, gradually replace them as the oxygen in the waste is depleted. Because of this shift in processes, there is a corresponding shift in gas composition. Carbon dioxide is produced in large quantities initially, and its production peaks in 2 weeks to 2 months, then falls until a stable plateau is achieved.[13] In contrast, methane may take from months to many years to reach peak production.

The rate of gas production increases with increasing moisture content and temperature of the solid waste; the production rate decreases with increasing density of the waste.[14] Other factors affecting gas production include the pH level, nutrient conditions, and metals content.

TABLE 3. SUMMARY OF COST ESTIMATES FOR LEACHATE TREATMENT[12]

	Leachate flowrate, liters/sec	Typical effluent COD, mg/liter		Costs of treatment, $/1000 liters leachate	
Influent BOD, mg/liter		25,000	5,000	25,000	50,000
Activated sludge (AS)	1.26	30	30	6.13	1.58
(Combined treatment)	0.13	30	30	10.92	3.14
Aerated lagoon (AL)	1.26	500	100	4.72	1.08
	0.13	500	100	8.34	2.64
Anaerobic filter (AF)	1.26	1500	300	5.83 (4.72)[a]	1.79 (1.55)
	0.13	1500	300	11.35 (10.24)	4.67 (4.43)
AL-sand filter (SF)	1.26	125	25	6.78	1.92
-activated carbon (AC)	0.13	125	25	10.53	3.61
AL-SF-AC-Reverse	1.26	25	5	7.28	2.42
Osmosis (RO)[b]	0.13	25	5	11.77	4.85
AF-SF-AC	1.26	375	75	8.65 (7.55)	2.79 (2.56)
	0.13	375	75	14.30 (13.2)	5.80 (5.57)
AF-SF-AC-RO[b]	1.26	75	15	9.16 (8.02)	3.3 (3.03)
	0.13	75	15	15.54 (14.33)	7.04 (6.70)

[a] Numbers in parenthesis indicate the costs of treatment after deducting credit for methane produced at $5.35/100 m^3.

[b] After RO treatment the total dissolved solids (TDS) decreased to 300 mg/liter and 60 mg/liter at influent leachate BOD concentrations of 25,000 mg/liter and 5,000 mg/liter, respectively.

Many and highly variable gas production rates have been reported;[14] the variation is mainly attributable to: (1) methods of data collection, (2) differences in theoretical assumptous and calculations, and (3) differences in the waste itself. Methane, carbon dioxide, hydrogen, and nitrogen are the major components of landfill gas. Minor components include argon, hydrogen sulfide, sulfides, disulfides, and an assortment of organic gases in trace amounts. Typical composition of landfill gas is presented in Table 4. As with production, the gas composition varies widely and is influenced by the same parameters.

Migration of Landfill Gas

Landfill gases may migrate considerable distances from the landfill through the surrounding soil. Because gas generation is not limited to the active life of the filling operation but continues for a period of years, gas control is a very important design consideration.

TABLE 4. RANGE OF LANDFILL GAS COMPOSITION[13,14]

(percent)

Methane	40-60
Carbon dioxide	30-60
Nitrogen	0-30
Hydrogen	<1
Oxygen	0-20

Gases migrate in all directions from the fill as a result of both pressure and diffusional flows. The gases move in the direction of decreasing pressure, and diffuse because of gradients in gas concentrations. In early attempts to model gas flow from a landfill the investigators recognized that the flow of gases around a landfill consists of a mixture of different gas types, but they accounted for this only indirectly. Recent work at the Ohio State University has been aimed toward more rigorous solutions to be incorporated into a user manual for design application. The work by Moore and others has provided techniques for predicting time-concentration profiles, computer programs for analyzing specific situations, design charts that may be used to estimate migration, and design criteria for control devices.[15]

Environmental Effects of Landfill Gas

The three main environmental effects associated with production of gas from decomposing solid waste are: the escape of odorous gases, fire and explosions due to accumulations of methane, and damage to vegetation on adjacent land. Although the major components of landfill gas are odorless, minor constituents such as hydrogen sulfide are extremely malodorous. Such gases can migrate from the fill and accumulate in structures, making them unpleasant and even dangerous to inhabit or work in.

In concentrations ranging from 5 to 15 percent by volume, methane is explosive in the presence of oxygen. Reports of fires and explosions resulting from accumulations of methane from landfills include incidents involving loss of life. Explosions in Georgia (1967) and North Carolina (1969) caused the deaths of five men.[13] These explosions were traced to the migration of methane from landfills and its subsequent accumulation in structures the men were occupying. In addition to the explosion hazard, methane also tends to displace oxygen and thus can cause asphyxiation.

A number of cases of adverse effects on vegetation caused by intrusion of landfill gases have been reported. Flower reported three cases in New Jersey involving injury and death of peach trees, ornamentals, and commercial crops.[16] These cases were associated with the decomposition of municipal solid waste deposited in worked-out sand and gravel pits, a situation that would tend to promote gas migration. Flower concluded in another report that soil characteristics, modified by landfill gas included moisture content and available ammonia-nitrogen, iron, manganese, zinc, and copper - all of which he found to be significantly increased by the presence of landfill gases.[17] Flower also found a high negative correlation between plant growth and concentrations of methane and/or carbon dioxide in the root atmosphere.

An additional environmental effect of landfill gas is the potential for carbon dioxide to dissolve in groundwater and thus to reduce the pH level through the formation of carbonic acid. Acidic groundwater is more susceptible to mineralization when it comes into contact with acid-soluble formations. The water may then be unfit for human consumption or for some industrial uses unless it is further treated.

Control of Landfill Gas

Vents and barriers are the two general types of control devices that can be used to control or prevent gas migration and thus protect nearby structures and vegetation. Vents provide an escape route for the gas, usually through trenches or pipes filled with gravel and open to the atmosphere. Barriers prohibit the movement of gas through an area limited by an impermeable material. Hybrid systems, which combine the mechanisms of vents and barriers, are also used. These gas control systems may be designed and installed before operation of the landfill or may be added on to control or alleviate a gas migration problem.

Pipe vents can relieve localized accumulations of gas in a sanitary landfill. Normally some type of convective flow must be introduced if pipe vents are to be effective. Likewise, a trench vent may be designed to vent naturally to the atmosphere or to undergo forced convection by mechanical pumping in either the exhaust or recharge mode.

A barrier can control gas migration only if it extends downward to another impervious material. Natural and synthetic impervious materials are used to line landfills to control gas migration. Both field experience and computer simulation suggest that passive vent systems (wells, trenches etc.) will relieve pressure but will not prevent migration due to concentration gradients.

Systematic monitoring of gas concentrations and movement is essential to determine the effectiveness of a control device and to ensure that gases do not accumulate to high levels.

Recovery and Utilization of Landfill Gas

Municipal solid waste disposal sites are untapped sources of methane gas. Landfill gas, which is mainly produced during anaerobic decomposition, has a volume composition of 40 to 60 percent methane. Other gases produced are carbon dioxide, nitrogen, and hydrogen sulfide. Table 5 shows the average gas composition obtained at the Mount View study site.

TABLE 5. AVERAGE GAS COMPOSITION AT MT. VIEW STUDY SITE[18]

Constituent	Volume %
Methane	44.0
Carbon dioxide	34.2
Nitrogen	20.8
Oxygen	1.0

The amount of recoverable gas depends upon two factors: the ultimate gas production and the area-depth relationship. Ultimate gas production values have ranged from 0.003 to 0.43 scm/kg of refuse.[19] These estimates include lysimeter studies conducted at optimal conditions and include the addition of sewage sludge to municipal refuse. It is estimated that 0.15 scm of raw landfill gas per kilograms of refuse will be obtained at Mt. View. Since this raw landfill gas is 44 percent methane, 0.07 scm of methane per kilogram of refuse will be obtained.

The other factor concerns the area-depth relationship. Because of the possibility of air infiltration, a deeper landfill is preferred to a shallow landfill, even though volume is constant. If air infiltrates into the landfill, methane production will cause or decline to a level where the methane content of the raw gas is too low for economic recovery. The Mt. View site is 12 meters deep and has had problems maintaining high pumping rates (5.6 m^3/min). Pumping has been reduced to 1.4 m^3/min and a stable methane content of 44 to 45 percent has been achieved. This problem has been directly attributed to the infiltration of air from the higher pumping rate. At the higher rate, landfill conditions went from anaerobic to aerobic decomposition and the volume of methane dropped from 44 to 30 percent. The landfill required 2 months before anaerobic conditions were reestablished.[20] Deeper landfills (30 to 42 meters) in the Los Angeles area do not appear to have this air infiltration problems.

The heating value of raw landfill gas is approximately 16.9 x 10^3 kJ/scm. Depending on final use, landfill gas may be needed to be upgraded. At the Mt. View project, the landfill gas will be compressed, dehydrated, and stripped of carbon dioxide to produce a gas with a heating value 24.4 x 10^3 kJ/scm. This gas will then be fed into a Pacific Gas and Electric Company main transmission line for distribution to residential and industrial customers.

Cost estimates for collection and treatment, indicates that landfill gas, while not presently competitive with natural gas or oil, is competitive with LNG or SNG. Thus, landfill gas can be an important supplemental or alternative fuel source.

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ANALYTICAL METHODS EVALUATION
FOR APPLICABILITY IN LEACHATE ANALYSIS

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ABSTRACT

This study conducted a round robin analysis of 10 leachate samples that were analysed by 32 laboratories located in the United States and Canada. The laboratories analysed for up to 28 constituents to include physical parameters, organics, anions and cations. The study found that the overall coefficients of variation varied between 32% for the chemical oxygen demand to as high as 210% for the cadmium determination. Significant differences were noted between results from colorimetric methods and titrimetric and physical methods. Results obtained with automated methods generally deviated from other methods. The average recovery of spiked parameters in one leachate sample was 81% but varied widely for individual parameters. These findings point to the necessity of using the standard addition technique to determine the matrix depression or enhancement for each leachate sample. Splitting samples between laboratories is also highly recommended as between laboratory variation is more significant than within laboratory variation.

INTRODUCTION

Municipal solid waste is primarily disposed of in sanitary landfills. While such a disposal method is optimum from a public health standpoint, it can have adverse effects on the environment unless sound engineering principles are used during design, operation and longterm maintenance of the solid waste fill. Infiltrating rainwater will dissolve organic and inorganic substances from the fill. This leachate may than migrate into the surrounding soil and pollute groundwaters and nearby aquifers. The assessment of the environmental impact of leachate requires the accurate and consistant determination of a substantial number of groundwater contaminants. An earlier study by Chian and DeWalle (1975) tested the standard addition technique for 29 parameters in a concentrated leachate sample. Several methods, especially those based on colorimetric methods, showed strong interferences due to the color and suspended solids present in the leachate. The present study conducted a round robin analysis of 10 leachate samples that were analysed by 32 laboratories for 28 constituents, to determine the analytical variability of each method.

DATA AND ANALYSIS

A total of 120 laboratories that were routinely conducting leachate analysis, were contacted to learn about their interest in participating in a round robin study. Fifty two laboratories indicated their willingness to participate, among which Federal and State laboratories were best represented (60%) followed by commercial (20%) and university (20%) laboratories. Large leachate or polluted ground water samples were subsequently collected at nine sites in the US and Canada. Four of these were full scale landfills, two were pilot scale landfills while three were laboratory scale solid waste lysimeters as shown in Table 1. The COD strength ranged from 98 to 27,284 mg/l. These values were close to the estimated value submitted to each laboratory ahead of time and therefore greatly aided in the selection of the right dilution. Thirty two of the fifty two laboratories completed the analysis of the

TABLE 1. LEACHATE COLLECTION LOCATIONS AND INITIAL ANALYTICAL RESULTS

Location	Well/Drain	Symbols	Estimated COD (mg/ℓ)	Measured COD (mg/ℓ)
Center Hill, Cincinnati, Ohio	Lysimeter Cell 16 Chlorine Brine Drainpipe	A	30,000	27,284
Enfield, Connecticut	Groundwater Well 26	B	30,000	2,168
Tellytown, Pennsylvania	Combined leachate Influent	C	23,000	16,084
Llangollen, Delaware	Groundwater Well S 1	D	10,000	10,719
Univ. of Illinois, Urbana, Illinois	Lysimeter Drainpipe (spiked)	E	7,500	3,408
Univ. of Illinois, Urbana, Illinois	Lysimeter Drainpipe	F	5,000	2,047
Boon County, Kentucky	Test Cell Upper Drainpipe	G	3,000	945
Univ. of British Columbia, Vancouver, British Columbia	Lysimeter Cell K Drainpipe	H	1,800	1,298
Sonoma County, California	Test Cell D Recycle Drainpipe	I	700	423
Dade County, Miami, Florida	Groundwater Well 4-20	J	100	98

10 leachate samples, although not all parameters were tested by all laboratories. The least reported parameters were the oxidation reduction potential (11 laboratories), reflecting the anaerobiosis of the sample and the volatile acids (12 laboratories) indicating the extend of the cellulose decomposition and subsequent methane fermentation.

The data were evaluated using the analysis of variance with the laboratories and samples as factors, both considerd to behave in a random fashion. This allows an estimate of the between laboratory variance σ_L^2, within laboratory variance σ_W^2, sample variance σ_S^2 and variance from laboratory-sample interaction σ_{LS}^2. In almost all instances a natural logarithmic transformation of the data was required to make the variability at different levels of the parameter approximately equal. The assumption of homogeneity of variance was tested by relating the within and in-between laboratory standard deviation to the concentration of the parameter in the leachates. The within laboratory standard deviation was calculated by pooling the standard deviations computed from the replicates of each laboratory, while the standard deviations between laboratories was computed for each leachate sample from the laboratory averages. Each standard deviation σ was related to the mean through: $\sigma = a.\mu^b$. If the estimated interval for the power term b contained 1, a $\log_e$ transformation was applied. The standard deviation of the data in the $\log_e$ transformed scale is approximately equal to the coefficient of variation of the data in the original scale as determined from a first order Taylor expansion of the transformed data. All

TABLE 2. SUMMARY OF ANALYTICAL RESULTS FOR COD DATA*

Leachate sample	Grand sample average	Minimum lab average	Maximum lab average	Median of lab averages	Within lab std. dev.	Between lab std. dev.
A	27284	14933	40121	28700	990	5669
B	2168	816	8737	1888	310	1592
C	16084	5900	24876	16208	531	3259
D	10719	5900	17283	10990	423	2318
E	3408	1400	5488	3833	146	1154
F	2047	1074	4314	1962	259	705
G	945	380	1792	895	53.1	365
H	1298	580	1860	1329	143	271
I	423	295	940	394.	22.3	123
J	98.4	57.3	193	84.5	7.76	33.2

*All entries in mg/ℓ

leachate parameters were processed in the above fashion. However for illustrative purposes only the COD analysis will be discussed.

ANALYSIS OF THE COD DATA

Data was received from 25 of the 46 laboratories who agreed to perform the test on the 10 leachate samples. The results in Table 2 indicate the range of COD values reported with the maximum value about 4.3 times larger than the minimum value. Since a value of the power term b included 1, a log transformation was indicated (Figure 1 and 2). The normal frequency distribution and log normal frequency distribution of the individual data points are shown in Figure 3 and 4. A linear regression through the quantile-quantile plots showed that the average R^2 increased from 82% to 85% and that the average standard deviation of R^2 decreased

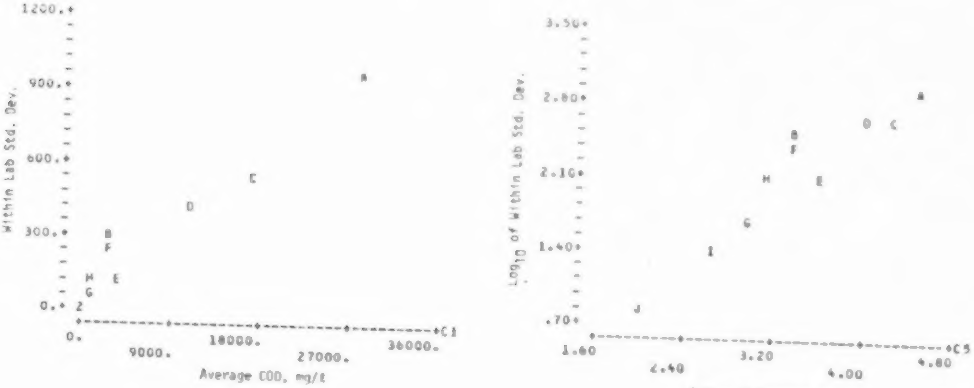

Figure 1. Relationship of the within laboratory standard deviation to the COD concentration level.

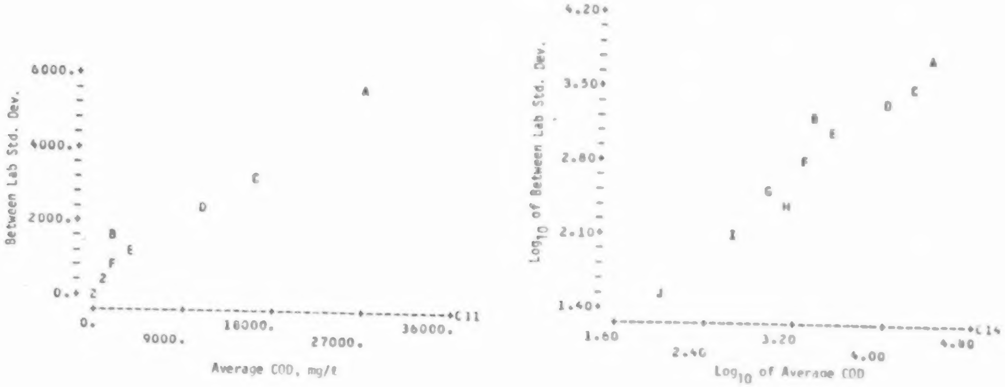

Figure 2. Relationship of the between laboratory standard deviation to the COD concentration level.

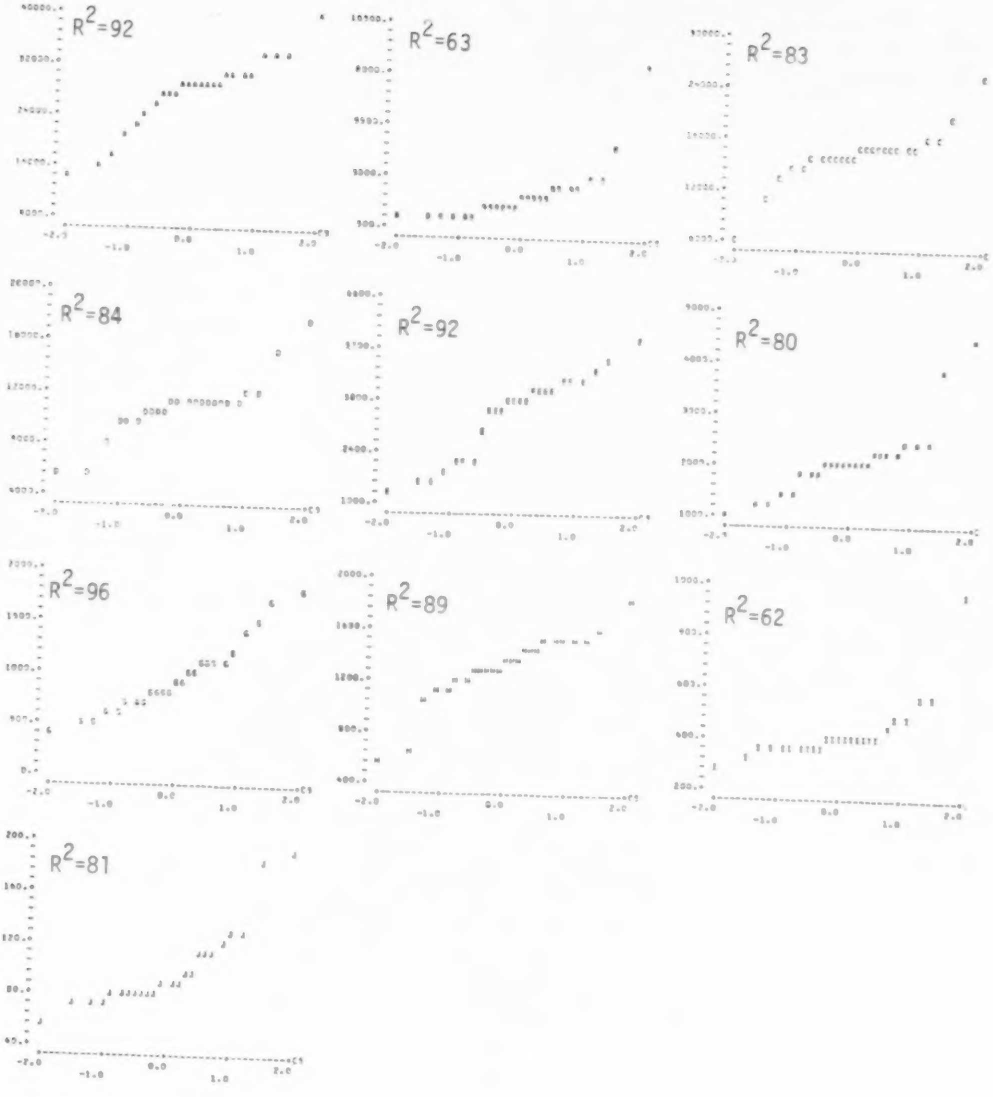

Figure 3. Ordered laboratory averages for COD data (vertical axis) plotted against quantiles from a normal distribution (horizontal axis) for leachate samples A through J respectively.

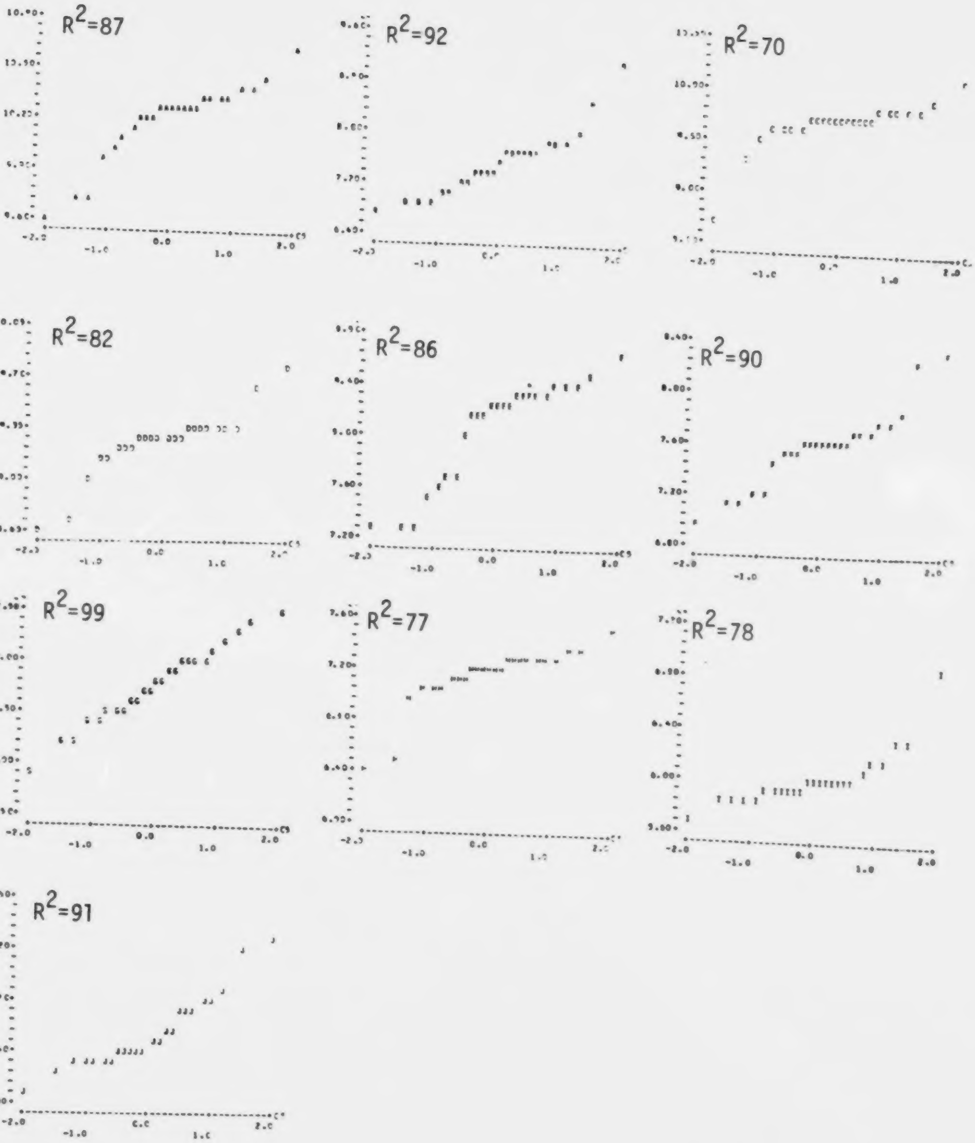

Figure 4. Ordered laboratory averages for $\log_e$ transformed COD data (vertical axis) plotted against quantiles from a normal distribution (horizontal axis) for leachate samples A through J respectively.

from 12% to 9% after the $\log_e$ transformation, indicating that the transformed data more closely resemble a normal distribution than the original data.

Most laboratories, i.e. 23 out of the 25 used the dichromate reflux method while one laboratory used the automated dichromate reflux method. One laboratory did not specify its method but probably used the former method. A simple non parametric test applicable to two-way classified data without interaction, the Friedman test, was used to determine whether laboratories using different methods had significantly different scores. The Friedman test consists of ranking the laboratories based on the average reported value for each leachate sample. The lowest value for analysis of a given leachate sample is assigned the rank of 1, the second lowest the rank of 2 etc. These ranks are then summed for each of the laboratories. The ranking for the two COD metods showed that the automated method had a score of 49, while the reflux method had a score of 132 indicating that the former tends to give substantially lower results than the reflux method.

The rank sum scored by each laboratory was used to determine whether excessive systematic error is attributed to those laboratories scoring low and high ranks. Assuming a chi square distribution,the 95% confidence interval was calculated. Testing showed that 4 laboratories had consistently low scores while 2 had high scores, indicating that 25% of the laboratories produce unacceptable results.

The analysis of variance on a balanced data set with three replicates showed an overall standard deviation, equal to the coefficient of variation in the original scale of 26.7%. The analysis of variance performed on the averages of all laboratories resulted in a coefficient of 32.4%. The largest component of the coefficient was made up of a laboratory-sample interaction indicating that some laboratories had consistently high or low values depending on the leachate sample. Comparable data generated in other studies (EPA, 1974) show a coefficient of variation of 6.6% for organic matter with a COD of 270 mg/l and 6.5% at 160 mg/l.

In none of the analysis was it possible to detect a time effect,i.e. a decrease of the COD due to bacterial degradation within the 40 day period that most analysis were conducted. Also no significant dilution effect was noted in which a higher COD is noted at a greater dilution due to a reduction of interfering substances.

ANALYSIS OF OTHER CONSTITUENTS

The analytical results showed an overall coefficient of variation ranging from 32% for the COD test to 210% for the cadmium determination as shown in Table 3. The average coefficient of variation for the physical parameters such as pH and conductivity, is 93%, for the organics such as COD and organic nitrogen, is 81%. The variation for the anions is 130% and for the cations 109%, indicating that the parameters such as sulfate and nitrate are the most difficult to determine precisely. The greatest difficulty in the metal analysis was noted for the cadmium and chromium analysis.

Different methods used for individual parameters tended to give variable results. Both the automated colorimetric COD method and the automated alkalinity determination tended to give lower results than the corresponding manual titrimetric methods. The values obtained with colorimetric Kjeldahl method for organic nitrogen, the automatic colorimetric method for ammonia nitrogen, the automated methylthymol blue method for sulfate and the automated ferricyanide method for chloride generally gave higher results than the corresponding manual methods.

Instrumental electrochemical methods also tended to give values that were different from the manual colorimetric or titrimetric methods. Both the ammonia and chloride determination with the specific ion electrode gave higher values than the corresponding manual methods.

Differences were also noted among instrumental methods. The barium determination, for example, gave higher results using the flame emission method than the atomic absorption alternative. The zinc determination with the atomic absorption graphite method tended to give higher results than the direct aspiration atomic

TABLE 3. VARIANCE COMPONENT ESTIMATES WITH VISUALLY OUTLYING LABORATORIES INCLUDED AND EXCLUDED*

Parameter	σ_R 1	σ_R 2	σ_L 1	σ_L 2	$\sqrt{\sigma^2_{LS} + \sigma^2_W}$ 1	$\sqrt{\sigma^2_{LS} + \sigma^2_W}$ 2	% of labs omitted
pH	.35	.14	.15	.10	.31	.10	24
ORP	123	75	93	39	80	64	36
Turbidity	2.00	1.30	1.50	.88	1.40	.95	23
Conductivity	.46	.14	.22	.11	.41	.10	21
Volatile Acids	1.00	.66	.63	.34	.79	.57	58
COD	.32	.24	.20	.12	.25	.20	28
TOC	.60	.28	.29	.16	.53	.24	27
Total Residue	.57	.21	.52	.13	.25	.16	11
Volatile Residue	.76	.29	.64	.18	.42	.22	26
Organic Nitrogen	1.50	.93	1.00	.76	1.10	.53	44
Ammonia Nitrogen	.89	.27	.72	.16	.52	.22	36
Sulfate	1.80	1.10	.60	.66	1.70	.90	61
Total Phosphorus	1.40	.63	1.20	.30	.82	.55	43
Chloride	.92	.17	.49	.03	.78	.16	30
Alkalinity	.38	.19	.22	.12	.31	.14	28
Nitrate Nitrogen	2.00	1.40	1.50	.90	1.30	1.10	39
Sodium	.54	.12	.12	.06	.53	.10	29
Potassium	.66	.16	.39	.12	.53	.11	17
Calcium	.74	.23	.64	.12	.37	.20	11
Magnesium	.59	.10	.11	.08	.58	.06	37
Barium	1.40	.67	1.00	.38	.94	.55	33
Iron	.53	.29	.28	.16	.45	.24	32
Zinc	1.40	.41	1.00	.18	.88	.37	31
Lead	.96	.79	.65	.51	.71	.60	30
Chromium	1.70	.70	1.30	.46	1.10	.53	36
Cadmium	2.10	1.00	1.80	.76	1.20	.64	37
Copper	1.40	.60	1.10	.31	.84	.51	28
Nickel	1.10	.57	.73	.30	.79	.48	32

* Turbidity through Nickel for $\log_e$ transformed data.
1 Visually outlying laboratories included.
2 Visually outlying laboratories excluded.

absorption method. On the other hand, the lead, chromium, nickel and cadmium as determined in the graphite method tended to give lower results than the direct aspiration method.

All parameters were also tested for time and dilution effects. Using a one sided t-test at the 5% significance level showed that no significant effect existed.

Since the UI leachate sample was send to the different laboratories both spiked and unspiked, it was possible to determine the recovery of the individual elements. While the average recovery was 81%, the value for the individual parameters ranged from -138% for lead to 328% for potassium (Table 4). In addition to lead negative recoveries were also noted for phosphate, iron and nitrate possibly indicating a precipitation reaction and occurence of denitrification.

CONCLUSIONS AND RECOMMENDATIONS

The present study noted that the between laboratory component of the variation was larger than the within laboratory variability. Substantial

TABLE 4. RESULTS OF RECOVERY STUDIES USING A SPIKED AND UNSPIKED LEACHATE SAMPLE

	Parameter	Spiked Sample E	Unspiked Sample F	Measured Increase of Spiked Parameters	Calculated Increase of Spiked Parameters	Percentage Recovery
1	pH	6.49	6.82	-0.33	-	-
2	ORP	-138	-196	-58	-	-
3	Turbidity	349	560	-211	-	-
4	Conductivity	5003	3185	1818	-	-
5	Fatty Acids	1514	497	1017	1062	95.7%
6	COD	3408	2047	1361	4136	32.9%
7	TOC	2220	660	1560	1141	137%
8	Total Residue	4155	2944	1211	4345	27.9%
9	Volatile Residue	2168	1425	743	2660	27.9%
10	Organic Nitrogen	168	107	61	35.5	172%
11	Ammonia Nitrogen	103	38.2	64.8	26.4	245%
12	Sulfate	112	29.5	82.5	156.5	52.7%
13	Total Phosphorus	6.42	6.83	-0.41	1.3	-31.5%
14	Chloride	275	107	168	97.7	172%
15	Alkalinity	1963	1701	262	145.5	180%
16	Nitrate Nitrogen	0.526	0.608	-0.82	0.65	-126%
17	Sodium	229	103	126	66	191%
18	Potassium	161	83	78	23.8	328%
19	Calcium	698	675	23	263.3	8.7%
20	Magnesium	99.6	60.7	38.9	35.5	109%
21	Barium	2.05	1.87	0.18	2.7	6.7%
22	Iron	161	172	-11	131.3	-8.4%
23	Zinc	5.41	4.67	0.74	6.3	11.7%
24	Lead	0.308	0.317	-0.09	0.065	-138%
25	Chromium	0.228	0.080	0.148	0.580	25.5%
26	Cadmium	0.033	0.010	0.022	0.007	314%
27	Copper	0.204	0.199	0.005	0.035	14.3%
28	Nickel	0.763	0.144	0.619	0.640	96.7%
29	Average					81%

differences were noted between different methods. Since more and more analysis in the future will be conducted using automated methods as opposed to manual methods, a detailed comparison between the two type of methods is required for several parameters. Such a comparison is also required between electrochemical methods using specific ion electrodes and the corresponding manual methods. An indepth comparison between the graphite AA and the direct aspiration AA is also necessary.

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DEVELOPMENT OF THE PROPOSED DEFINITION OF HAZARDOUS WASTE

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ABSTRACT

The Resource Conservation and Recovery Act of 1976 created a regulatory framework to control the disposal of waste materials. The Act divides wastes into two classes for control purposes. This paper summarizes how the broad legislative definition of a hazardous waste was translated into the one proposed on December 8, 1978 (Federal Register) for use in evaluating specific waste materials. The rationale for incorporating ignitability, corrosivity, reactivity, toxicity, radioactivity, and infectiousness as characteristics of a hazardous waste are discussed.

INTRODUCTION

The Resource Conservation and Recovery Act of 1976 (RCRA) created a regulatory framework to control the disposal of those wastes not adequately controlled by existing environmental legislation. The Act sets up two classes of waste. For hazardous waste, it creates a management control system which requires "cradle-to-grave" cognizance including monitoring, record keeping, and reporting throughout the system. To paraphrase the Act, hazardous wastes are those which may pose a substantial present or potential hazard to human health or the environment when improperly managed. All other wastes are considered to be non-hazardous and are subject only to requirements of the States. The Act calls for the prohibition of open dumping and requires environmentally acceptable practices. However, while EPA was given authority to write rules for what constitutes environmentally acceptable disposal, only in the case of hazardous wastes was authority given to enforce the rules. The Act also gives the States the option of administering the hazardous waste program as long as their program is at least as stringent as the Federal program. Thus, the Federal regulations will be the least stringent to which generators, storers, transporters, treaters, and disposers of hazardous waste must respond.

This morning I will discuss some of the highlights of how this broad definition was refined into one which can be used to determine whether or not a particular waste is hazardous. Six criteria were used in determining specific characteristics to use in defining a hazardous waste. These criteria were:

1. The particular characteristic was spelled out in RCRA.
2. We had information on incidents which showed a particular characteristic has caused damage.
3. Other government or private organizations which regulated or recommend management methods for hazardous substances have identified a characteristic to be of concern.
4. The characteristic could provide a general description of the hazard rather than appearing merely as a list of sources.
5. The likelihood of a hazard developing if the waste was mismanaged is sufficiently great, and,

6. A reliable identification or test method for the presence of the characteristic is available.

Use of this last criterion has led EPA to try and describe each characteristic by specific testing protocols. Where this was not possible, the characteristic was either set out as a prose description readily recognizable by persons working in the field, or reserved for future regulatory action.

It must be emphasized that neither the set of characteristics nor the specific definitions of the characteristics are static. Both may be added to or changed as new information develops.

Based on those previous criteria, the following hazard characteristics were considered:

1. Ignitability
2. Corrosivity
3. Reactivity
4. Toxicity
5. Radioactivity
6. Infectiousness

In order to provide specific descriptions of wastes meeting these characteristics, each characteristic was defined in terms of specific definable properties. The following is a brief description of each characteristic and its properties. It should be noted here that radioactivity and infectiousness are not included as characteristics at this time.

Ignitability

The objective of the ignitability characteristic is to identify wastes which present a fire hazard under routine disposal or storage conditions. The resulting fires present not only the immediate danger of heat and smoke, but also provide a pathway by which toxic particulates can spread to the surrounding area.

There are several methods which can be used to identify ignitable materials depending on the physical state. For liquid wastes, flash point was selected as the property to use. The specific value chosen, 60°C (140°F) was selected after considering the ambient temperatures to which waste may be exposed during management.

For solid materials, a descriptive definition was selected because of lack of suitable test methods. For waste gases, EPA proposed to use the DOT identification since the major hazard arising from flammable gases would be during transport.

Corrosivity

A corrosivity characteristic has been included for two reasons. First, to identify those wastes which need to be segregated or specially handled because of their ability to mobilize heavy metals which might otherwise not migrate. Secondly, to identify those wastes requiring special containers during transportation and storage.

While heavy metal solubilization is an extremely complex phenomenon, pH has been found to be its most important indicator. The pH limits chosen in these proposed regulations were based upon skin corrosion limits and heavy metal solubilization data. The metal corrosion limits were taken from DOT Hazardous Materials regulations, because EPA's concern about container damage is identical to that of DOT's in this case.

Reactivity

The objective of the reactive waste characteristic is to identify waste which under routine management presents a hazard because of instability or extreme reactivity. Reactivity includes the tendency to autopolymerize; to create a vigorous reaction with air or water; to exhibit shock and thermal instability; to generate toxic gases, and to explode.

The largest stumbling block to developing general test methods for use in identifying reactive waste is that while there are many inputs of energy that may cause a waste to react or exhibit hazardous properties, there is no one stress that can cause all reactive waste to do so. To compound the problem, reactivity is not just a function of the composition, temperature, and availability of initiating agents, but is also affected by the mass and geometry of the waste. Thus, the reactivity of a tested waste sample may not necessarily correspond to the reactivity of the waste as a whole.

Since reactive waste is dangerous to the generator's own operations (as well as being hazardous for long term disposal), generators of reactive waste tend to be aware that their waste has that characteristic.

For this reason, EPA feels that the proposed descriptive definition will be an adequate identification method when used in conjunction with the test methods identifying thermal and shock instability.

Toxicity

The toxicity characteristic is intended to identify waste which, if improperly disposed of, may release toxicants in sufficient amounts to pose a substantial hazard to human health or the environment. The RCRA definition of hazardous waste requires EPA to make a judgement as to the hazard posed by a waste "when improperly treated, stored, transported, or disposed of, or otherwise managed." For waste containing toxic constituents, this hazard is dependent on two factors:

- the intrinsic hazard of the constituents of the waste, and
- the release of the constituents to the environment under conditions of improper management.

To assess the intrinsic hazard posed by the constituents a series of toxicity indicators has been considered. Those which we feel are of sufficient concern to be eventually incorporated in a hazardous waste definition are:

- acute and chronic toxicity to humans, animals, and plants,
- potential for bioaccumulation in tissue,
- oncogenicity,
- mutagenicity, and
- teratogenicity.

However, the toxicity definition proposed in December 18, 1978, is limited to the onset of toxicants for which National Interim Primary Drinking Water Standards (NIPDWS) have been developed. To determine whether toxic constituents in the waste might migrate in the disposal environment, a procedure has been developed to measure the tendency of the constituents of a waste to leak or leach out and become available to the environment under poor management conditions. This procedure has been termed the Extraction Procedure and will be discussed later.

Radioactivity and Infectivity

Two other properties of wastes that were deemed to be of concern are radioactivity and infectiousness. After studying this area, it was decided that sources and types of wastes posing these hazards are relatively few. It was thus decided that, at this time, instead of specifying specific tests for these characteristics, identification would be via a listing of sources.

As the toxicity characteristic is one of the more complicated and far reaching aspects of the definition, its development will be discussed in more detail.

Numerous studies and reports indicate that damage to ground and surface water frequently results from migration of toxic chemicals from the initial disposal site. Groundwater contamination is a particularly important concern because it is a source of drinking water for almost half of the population. In addition, once contaminated, its usefulness as a source of drinking water may be impaired for years. It was thus decided that use of a groundwater contamination scenario to "model" improper disposal would be advisable. By selecting a groundwater contamination scenario, we did not mean to imply that other vectors are not important. However, we do feel, that except in rare cases, control levels set using this model will be sufficient to protect against other routes of contamination.

The model is based on wastes creating a problem through migration of chemicals out of the disposal site and into a drinking water aquifer. I want to emphasize that the contamination model has been developed for definitional purposes only. It does not address actual disposal methods which might need handling in a manner different from ordinary refuse.

Once the specific properties of toxicity were selected, other objectives became important. These include:

1. Formulation of a dynamic definition which would not only identify those wastes which contain known toxicants, but also those wastes which contain materials or combinations of materials whose toxic properties have not yet been recognized, and

2. specification of toxicant control levels consistent with environmental goals formulated under other regulatory authorities.

In order to take into account the difficulty of formulating a test scheme applicable to wastes of widely varying complexity, it was decided that having definitions based on both analytical and biological parameters would be desirable. Either one could be used to evaluate whether a given waste is hazardous. The analytical set would rely on a quantitative analysis of an extract of the waste, combined with hazard thresholds based on mammalian, aquatic, and terrestrial plant toxicities. If the concentration of any species exceeded the calculated threshold value, then the waste is deemed to be a hazardous waste. A bioassay approach would also be available to use when complicated or hard to analyze wastes are to be evaluated. In this approach, sensitive aquatic and plant species are exposed to an extract of the waste and examined for signs of toxicity. If manifestations of toxicity are noted, then the waste is a hazardous waste. We feel that this type of definition is desirable for several reasons:

1. By keying to waste properties, rather than a static list of known hazardous materials or wastes, the definition would be dynamic. As new toxic agents enter the waste disposal network they would then be immediately covered.

2. Using biological indicators offers a mechanism for assessing toxicant synergism and antagonism in complex mixtures characteristic of wastes.

3. Testing costs are low enough so that non-hazardous wastes will not be forced into the hazardous waste net as a result of prohibitively expensive test procedures.

The test scheme devised to meet the total definition of toxic waste is outlined in Figure 1. However, at this time, it must be pointed out that the characteristic defining toxicity has been severely limited in the regulations proposed on December 8, 1978. The proposed definition is limited to those wastes containing significant amounts of leachable substances for which a National Interim Primary Drinking Water Standard has been set.

The complete definition would employ biological tests for mutagenic activity and environmental persistence coupled with an instrumental method for bioaccumulation potential. It includes a choice between using either bioassay or analytical tests for measuring chronic toxicity, aquatic toxicity, and terrestrial plant toxicity. For those toxicants known to be either mutagenic, carcinogenic, or teratogenic, but which are not biologically active in the *in vitro* mutagenicity assays prescribed, control will be via listing on the "Controlled Substances List." In order to keep testing costs to a minimum, consistent with the need for adequate information, short-term *in vitro* bioassays have been selected whenever possible.

However, while we believe that the toxicity definition that I have just outlined would substantially meet the goals previously described, we feel that use of this definition is premature at this time. Three factors accounted for this decision.

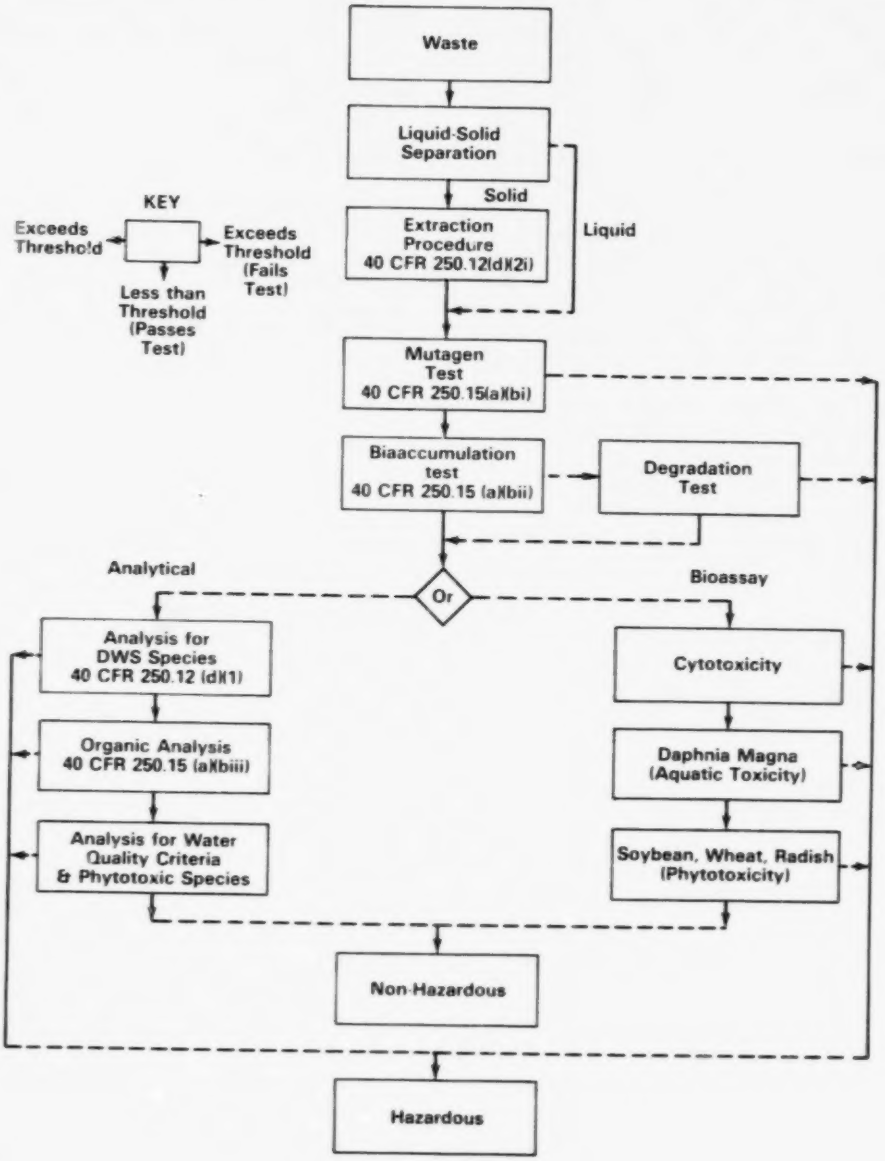
1. We have not had sufficient time to adequately validate and debug the bioassay procedures.

2. Concern over using the NIOSH Registry data in a manner for which it was not intended.

3. The lack of adequate data with which to determine the size of the hazardous waste class which would be so created.

Thus, in order to carry out the mandate of RCRA and implement a hazardous waste control program without further delay, a modified approach was proposed on December 18th. As I mentioned earlier, this approach makes use of the Extraction Procedure to measure toxicant availability combined with use of EPA National Interim Primary Drinking Water Regulations as a basis for determining threshold levels. However, since many wastes contain mobile toxic chemicals for which there are no drinking water standards, an expanded use has been made of lists as identifiers of hazardous waste. These hazardous waste lists contain those industrial processes and wastes for which the Agency has sufficient information to indicate that, in general, disposal of the waste requires controlled management. In order to give the generator of a listed waste a means of demonstrating that his particular waste

FIGURE 1
Protocol for classification of hazardous wastes


KEY
Exceeds Threshold
Exceeds Threshold (Fails Test)
Less than Threshold (Passes Test)
Waste
Liquid-Solid Separation
Solid
Liquid
Extraction Procedure 40 CFR 250.12(d)(2i)
Mutagen Test 40 CFR 250.15(a)(bi)
Biaaccumulation test 40 CFR 250.15 (a)(bii)
Degradation Test
Analytical
Or
Bioassay
Analysis for DWS Species 40 CFR 250.12 (d)(1)
Organic Analysis 40 CFR 250.15 (a)(biii)
Analysis for Water Quality Criteria & Phytotoxic Species
Cytotoxicity
Daphnia Magna (Aquatic Toxicity)
Soybean, Wheat, Radish (Phytotoxicity)
Non-Hazardous
Hazardous

is in fact not hazardous, test procedures have been developed.

While use of these analytical, mutagenic, and bioaccumulative tests have not been included in the hazardous waste definition because of a lack of data on their impact, this is not a problem when they are used for delisting purposes. In order for a generator to demonstrate that his waste is not hazardous, the Agency will accept data using these tests.

VEGETATION GROWTH IN LANDFILL ENVIRONS

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ABSTRACT

During the past dozen years, many attempts to revegetate completed sanitary landfills have been undertaken throughout the United States, with variable degrees of success. This has been evaluated in a recent nationwide field survey of vegetation growth on completed sanitary landfills. Based on the results of this survey, literature reviews and other field experiences, a study was undertaken to determine which species, if any, can maintain themselves in a landfill environment; to investigate the feasibility of preventing landfill gas from penetrating the root zone of selected species by using gas barrier techniques; and to identify the (those) factor (s) which are most important in maintaining adequate plant growth on completed sanitary landfills. Ten replicates of nineteen woody species were planted on a ten-year old completed sanitary landfill and five gas barrier systems were constructed. The experiment was replicated on nearby old forest land to act as a control. Of the nineteen species planted on the landfill for the past two years, certain species have tolerated the landfill conditions better than others. Where the gas barrier technique kept landfill gases from the root zone the trees grew best.

Carbon dioxide and methane are the major components of sanitary refuse landfill-generated gas which have been associated with the demise of vegetation on and adjacent to completed landfills. An investigation of the effects of carbon dioxide (CO_2) and/or methane (CH_4) contaminated soil atmospheres on the growth of tomato (Lycopersicon esculentum) plants indicated that CO_2 per se was toxic to tomato roots in a low O_2 soil atmosphere, whereas CH_4 per se was innocuous under the same conditions. Investigations into the effects of CO_2 and CH_4 contaminated soil indicated that red maple (Acer rubrum) is more tolerant to the presence of these gases than is sugar maple (Acer saccharum).

INTRODUCTION

The pressures of population expansion and urbanization have prompted a reappraisal of anticipated uses for completed landfill sites. Conversion to recreational areas or other non-structural usage has been considered an acceptable end for completed landfill sites in urban areas; and in rural areas intensifying land use has resulted in attempts to use completed landfills for growing commercial crops. Numerous farmers, as well as scores of landscapers, have encountered mixed success in trying to establish agricultural crops, trees, and shrubs on landfills throughout the country. Three questions are often raised: "What species will thrive on completed landfill sites?", "Are there any techniques available which will help in attempting to establish a vegetative cover over a completed landfill area?", and "What is the nature of the toxic effect of landfill gas on vegetation?"

Reports from a Rutgers University (New Brunswick, New Jersey) nationwide mail survey funded by the Federal E.P.A. Solid and Hazardous Waste Division determined that the scope of problems encountered when vegetating completed landfills was indeed of national latitude (15). It was ascertained, from on-site visits to some 60 vegetated landfills, that answers to the previously raised questions would benefit not only the landscaper or farmer trying to vegetate a former landfill, but the general public as well in that they too would ultimately derive value from successful vegetation projects such as parks, golf courses and recreational areas.

In order to investigate the possibility of successfully growing vegetation on such areas, two experiments were designed: (1) a field experiment with three objectives: (a) to determine the relative tolerance of a number of commonly grown tree and shrub species to the soil environment created on and adjacent to a sanitary refuse landfill; (b) to determine if barriers to the migration of decompositional gases can function in preventing gas contamination of the root systems of selected species; (c) to identify those soil factors which are most responsible for causing vegetation growth problems on completed landfills. (2) A greenhouse experiment to assess the effects on vegetation of soil contamination by simulated landfill gas (CO_2 and CH_4) mixtures.

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LITERATURE REVIEW

Serious disadvantages for adequate vegetation growth inherent in landfill sites; namely the production of toxic gas mixtures from anaerobic decomposition of organic matter present, as well as high ground temperatures, have been reported (38).

The composition of landfilled refuse varies considerably depending on its origin, be it municipal, industrial, incineration ash or sewage sludge. The organic content of solid waste collected from homes, schools, commercial establishments and industries generally ranges from 50 to 75% on a weight basis. Most of these organics are biodegradable and can be broken down into simpler compounds by both aerobic and anaerobic organisms. The rate at which this occurs is reported to be a function of a variety of factors (8, 28).

When the refuse is initially deposited in the landfill, there is enough oxygen present to support a population of aerobic bacteria. This state lasts from one day to many months (12). The literature indicates carbon dioxide and water to be the principal products formed in aerobic decomposition (6). After the oxygen concentration is depleted, the aerobic bacteria die, and there is a sharp increase in the anaerobic bacterial population. During anaerobic decomposition CO_2 and CH_4 are the principle gases produced (11, 35). Various other gases reportedly produced in the anaerobic environment of the landfill include ethane, propane, hydrogen sulfide, nitrogen and nitrous oxide (1, 7, 26, 35). Reserve Synthetic Fuel Company reports finding over 60 different gases in a California landfill (18).

In addition to the methane-producing bacteria mentioned above, there exists a bacterium, Pseudomonas chromobacterium, which utilizes methane during its metabolism. It oxidizes methane, producing carbon dioxide and water (22). Since oxygen is required for this reaction, these bacteria will generally be found near the upper surface of the landfill and in lands adjacent to the landfill.

In spite of predictable negative success in utilizing landfill for the support of vegetation, many reports of success or proposals for transforming barren former refuse sites into luxuriant vegetated areas have appeared in the literature and in the press (2, 4, 22, 23, 29). In July 1972, an article by Duane (9) applauding the construction of golf courses on completed sanitary landfills cited the successful use of such tree species as Japanese black pine, London plane, thornless honey locust and Russian olive for beautifying the sites. In 1973, an anonymous article appeared in Solid Waste Management magazine describing the transformation "From Refuse Heap to Botanic Garden", of an 87-acre landfill in Los Angeles that had the distinction of being the world's first such phenomena (3). Few problems if any were either observed or anticipated in achieving these spectacular results with the exception of the report of root damage to large trees and shrubs at the Los Angeles Botanic Garden site.

At the same time, various investigators were experiencing difficulties in growing vegetation at similar sites. In January 1969, Professor F. Flower and associates of Rutgers University in New Brunswick, New Jersey (17), responding to a complaint of vegetation death on private properties adjacent to a landfill in Cherry Hill Township observed dead trees and shrubs of the following species: spruce, rhododendron, Japanese yew, azalea, dogwood, and others. Testing of the soil with appropriate equipment disclosed high concentrations of carbon dioxide and explosive gases. The conclusion reached was that the trees and shrubs might have been killed by displacement of oxygen from their root zones by lateral movement of the gases of refuse decomposition or by the decomposition gases themselves. Additional vegetation deaths caused by the migration of landfill gases were first noted at the Hunter Farm in Cinnaminson, New Jersey in 1970 and at a peach orchard in Glassboro, New Jersey in 1971 (14).

On a December 1974 visit to the Hunter Farm, when fields planted with rye were growing poorly (16), gas checks revealed that combustible gases were present in the area of new vegetation injury and that migrating gases were reaching up to 600 feet from the nearest edge of the landfill. Another trip to Hunter's Farm was made in June 1975 when corn was found to be growing poorly in areas where combustible gas and CO_2 concentrations were high (14).

In 1945, Chang and Loomis conducted a general survey of the literature and concluded that plants would survive concentrations of 1 to 2% oxygen in the root zone. They also concluded that most plants should function normally at soil oxygen concentrations ranging from 5 to 10% (7). Dense soil also increases the need for oxygen at the growing root tip. This is believed to be due to the extra work that has to be done by the root tips as they push their way through the soil (21). Higher temperatures were found to increase the oxygen requirement for growing roots (32).

Low concentrations of oxygen have been reported in the soil near natural gas leaks (19). A similar situation has been found both on and adjacent to sanitary landfills. Oxygen volume concentrations in the soil on landfills were found to range from 1% to 20% of the soil atmosphere (14).

There has been a considerable amount of research in establishing tolerance among different species to excess carbon dioxide in the root zone. Cotton seedlings grown in hydroponic solutions were able to make optimum growth with 10% carbon dioxide present, provided at least 7.5% oxygen was also present. Thirty to 45% carbon dioxide, however, was found to severely reduce root growth (25). Red and black raspberries were killed when their roots were exposed to 10% carbon dioxide. Root growth in broad bean and kidney bean was completely inhibited by 5.5% carbon dioxide (32).

Volume concentrations of soil carbon dioxide in the cover of landfills visited ranged from less than 1% to 42% of the soil atmosphere. A large percentage of these readings were in the 5% to 15% range (15). Normal soil CO_2 concentrations range from 0.04% to 2% (25); therefore, the levels recorded in the survey may be considered excessive.

There is convincing indirect evidence that shoot growth in trees is related to water deficiency, another stress characteristic of landfill cover soil. Forest mensurationists find tree height the most sensitive growth parameter for measuring site productivity, to which the generally accepted key is soil moisture (33, 34, 37). Correlations between rainfall and shoot growth have been attempted with various degrees of success for many decades (27, 30), and there is little doubt that wet years produce longer shoots than dry years for many tree species on upland sites. Root tissues are probably never at as severe water stress as shoot tissue because roots are closer to the water supply than leaves.

Flood tolerance may be related to CO_2-O_2 ratios in the soil. Hook et al (23) investigated flood tolerance in swamp tupelo and concluded that under flooding conditions, this species develops new roots that accelerate anaerobic respiration rate in the absence of oxygen, oxidize their rhizosphere, and tolerate high levels of carbon dioxide. The absence of any of these root adaptations would appear to reduce flood tolerance of a species. White (37) observed survival of certain species under water for 10 days to 3 weeks after hurricane Agnes in 1972. Those which did not tolerate flooding conditions included Norway maple, sugar maple, white birch, gray birch and many others.

PROCEDURES AND METHODS

Field Studies

An ideal site for a landfill study was defined as one which has a relatively high combustible gas concentration and adequate drainage. The control site should be located on virgin land, close to, but not adjacent to the landfill and have no combustible gas in its soil atmosphere. The sites should be large enough to accomodate a 4' spacing between adjacent trees.

The Edgeboro Landfill, on which the experiment is being conducted, is located on a marsh adjacent to the Raritan River. The deposited refuse is reported by the landfill owner to be approximately 30' deep under the experimental plot. Some methane may also be generated in the marsh beneath the refuse. The general municipal refuse landfill was completed at this location early in 1966. Later that year, 6"-10" of soil were placed over the refuse as a final cover.

The experimental plot is 72' x 108', an area large enough to accommodate the screening of 10 replicates of 19 different tree and shrub species and 5 landfill gas interception systems. The control plot is located approximately a quarter of a mile from the experimental plot and is on a former undisturbed woodland. It is 46' x 110' and accommodates 10 replicates of the same 19 species planted on the experimental plot in addition to 2 gas-barrier systems. Enough soil was spread on the experimental landfill plot to bring the total depth of cover soil to 2'. Two feet of the same soil was spread on the control plot as well.

Following the designation of the experimental plot, the precise boundaries for 5 gas interception systems (three 10' x 14' trenches and two 14' x 18' mounds) were determined by selecting the areas of highest combustible gas concentration in the soil atmosphere. A catepillar tractor bulldozer was used to dig the three 3' deep trenches and to move the excavated rubbish from the experimental site.

In order to prevent landfill decomposition gases from penetrating the soil, the 3 trenches were lined at the bottom with various barrier materials prior to backfilling with topsoil (Figure 1A). One trench was underlain with a one-foot deep layer of 1" round road-gravel beneath a sheet of 4-mil polyethylene plastic. Ten five-foot long, four-inch diameter perforated P.V.C. pipes were placed through the plastic and into the gravel equispaced around the trench. A second trench was lined on the bottom with a 1-ft. thick layer of clay and also contained 10 vertical vent pipes located around the periphery of the trench. The third trench was underlain with a 1-ft. thick layer of clay and contained no vents. Topsoil (79% sand, 12% silt, 9% clay) was then backfilled into each of the trenches.

Two 3-ft. high soil mounds were constructed using the same soil as in the trenches. One was underlain with a 1-ft. thick layer of the same clay used in the trenches while the other mound contained no gas barrier (Figure 1B).

In order to screen species representative of a maximum number of desirable landscaping characteristics and a variety of genotypes, plants were chosen from a number of categories (Table 1). Nineteen species were selected on the basis of these eight criteria and planted in replicates of 10 on the experimental landfill and control screening areas totaling 380 plants. Six American basswood and 4 Japanese yew were chosen for planting on each of the trenches and mounds because of the reported susceptibility of these species to landfill gases.

In order to characterize the landfill cover-soil environment, gas samples were collected from forty-eight buried samplers approximately every two weeks, beginning in March and ending in August 1977. Forty-two of the sampling stations were on the experimental plot and six on the control plot. Gas samples were collected from the samplers by means of gas-tight syringes and analyzed for O_2, CO_2, N_2, and CH_4 in a Carle Instrument Model 8500 Gas Chromatograph. Soil temperatures at the 1' depth were recorded from the same sample point and on the same dates as the gas samples.

Beginning in mid-March, 1977, soil moisture measurements were made at two week intervals on six samples from the experimental and four from the control screening area, and one from each of the gas interception systems. Soils were measured for bulk density in each planting

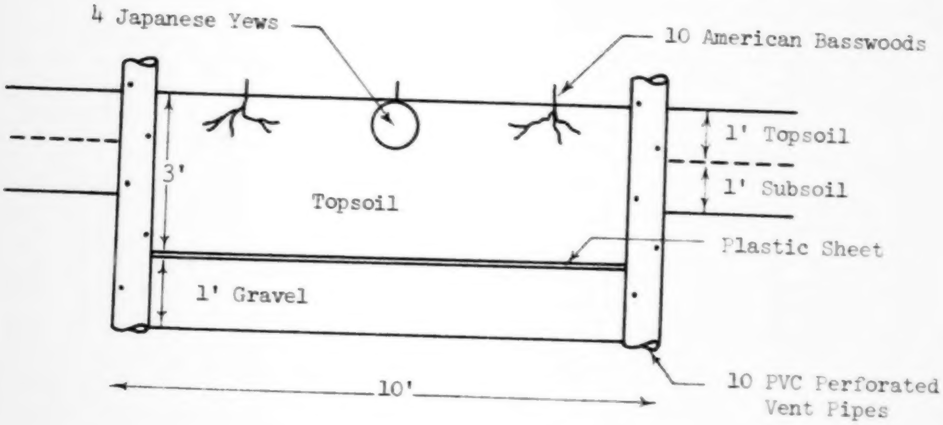
4 Japanese Yews
10 American Basswoods
1' Topsoil
1' Subsoil
3'
Topsoil
Plastic Sheet
1' Gravel
10'
10 PVC Perforated Vent Pipes

Figure 1A. Cross section end view of gravel/plastic/vents trench

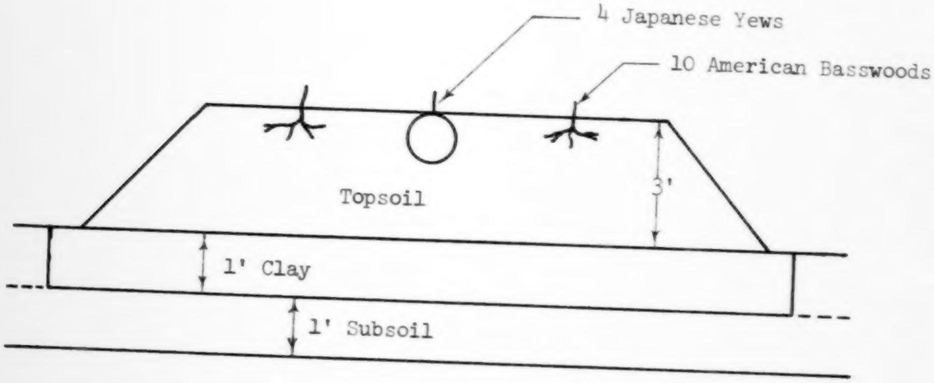
4 Japanese Yews
10 American Basswoods
Topsoil
3'
1' Clay
1' Subsoil

Figure 1B. Cross section end view of soil mound

TABLE 1. TREE PLANTING SELECTION CRITERIA

	Trees	Tolerance to Low O_2 Tension Environments	Aesthetic Landscaping Purposes	Other * Criteria
DECIDUOUS TREES	Honey Locust (Gleditsia triacanthus)	X	X	
	American Sycamore (Platanus occidentalis)	X	X	3
	Red Maple (Acer rubrum)	X	X	1
	Green Ash (Fraxinus pennsylvancia)	X	X	
	Black Gum (Nyssa sylvatica)	X	X	
	Weeping Willow (Salix babylonica)	X	X	
	Pin Oak (Quercus palustris)	X	X	
	Ginkgo (Ginkgo biloba)		X	3
	Sweet Gum (Liquidambar styraciflua)		X	1
	American Basswood (Tilia americana)		X	4
	Hybrid Poplar (Populus sp.)		X	
	Mixed Hybrid Poplar (Populus sp.)		X	
DECIDUOUS SHRUBS	Euonymus (Euonymus alatus)		X	
	Bayberry (Myrica pennsylvanica)		X	
BROADLEAF EVERGREEN SHRUB	Rhododendron (Rhododendron elegans)		X	
NEEDLE LEAF EVERGREENS	White Pine (Pinus strobus)		X	
	Japanese Black Pine (Pinus thunbergi)		X	2
	Norway Spruce (Picea excelsa)		X	
	Japanese Yew (Taxus cuspidata capitata)		X	4

* Other Criteria

1 Ubiquity
2 Seal salt tolerance
3 Tolerance to city conditions
4 Susceptibility to landfill conditions

technique and in both the experimental and control screening area.

In order to characterize the growth of each tree on the experimental and control plots during 1976, four measurements of shoot length were obtained for each tree. During the 1977 growing season, one measurement each of root biomass and cross sectional stem basal area, four measurements of leaf weight and six measurements of shoot length were obtained for each tree.

Simulated Landfill Studies

In order to select realistic concentrations of landfill gas components for inclusion in simulated mixtures for greenhouse studies, soil gas concentrations of 19 sanitary landfills visited throughout the continental United States were measured between August 1975 and January 1977 (16).

To investigate the ability of tree seedlings to survive in soil atmospheres contaminated with excessive amounts of carbon dioxide (CO_2) and methane (CH_4), two species of maple were chosen: red maple (Acer rubrum) because of its ability and sugar maple (Acer saccharum) because of its inability to withstand flooding. These species were compared in order to determine if the species more tolerant to flooding was also more tolerant to soil atmospheric contamination with CO_2 and CH_4.

One-year old red and sugar maple seedlings were planted (5 each) in twelve 20-gallon modified galvanized steel trash cans (Figure 2).

The seedlings were divided into three treatments with 20 seedlings (10 of each species) in each treatment. For treatment 1 the soil in each can was fumigated with a gas mixture containing approximately 3% O_2, 40% CO_2, 50% CH_4 and 7% N_2. Treatment 2 was a control with compressed ambient air forced through the soil. In treatment 3 the seedlings were flooded by filling the cans with water to a depth of 2 inches. In order to fumigate the soil, two cans were attached in parallel to a cylinder containing the gas mixture.

The composition of the soil atmospheres was monitored by extracting a 0.5 ml. sample of air from samplers buried at a 7 inch depth in the soil and analyzing it with a Carle model 8500 gas chromatograph.

The physiological condition of the seedlings was monitored by periodically measuring the rate of transpiration with a Lambda Instruments Diffusive Resistance Porometer. This instrument measures water vapor that evaporates from the leaf surface. If the root system was damaged, the tree would be unable to take up water fast enough to support normal transpiration.

In order to investigate the effect of simulated landfill gas on tomato plants, Rutgers tomato (Lycopersicon esculentum) plants were grown in specialized 4-liter culture vessels in sand solution culture (Figure 3).

When the plants were about 1 foot high, the lower leaves were pruned and lids were placed on the glass vessels (Figure 3). Cotton impregnated with heavy duty silicon vacuum grease and vaseline was used to provide a seal around the stems of the plants.

Tomato experiment 1 was designed to examine the response of tomato plants to a soil atmosphere having suppressed O_2 concentrations in conjunction with elevated concentrations of CO_2 and CH_4 (Table 2). This experiment consisted of 21 plants which were divided into 3 treatments with 7 replicates for each.

TABLE 2. COMPOSITION OF ATMOSPHERES USED TO FUMIGATE TOMATO PLANTS IN EXPERIMENT 1

	TREATMENT		
	A*	B	C
$\%O_2$	21	7	7
$\%CO_2$	trace	trace	10
$\%N_2$	79	93	58
$\%CH_4$	0	0	25

* Atmospheric air

Experiment 2 was designed to determine the effects of a soil atmosphere containing high concentrations of both methane and carbon dioxide on the growth of tomato plants compared with soil atmospheres containing only carbon dioxide or methane (Table 3). This experiment consisted of 20 plants which were divided into 5 treatments with 4 replicates for each.

TABLE 3. COMPOSITION OF ATMOSPHERES USED TO FUMIGATE TOMATO PLANTS IN EXPERIMENT 2

	TREATMENT				
	A*	B	C	D	E
$\%O_2$	21	5	5	5	5
$\%CO_2$	trace	trace	40	40	trace
$\%N_2$	79	95	55	5	45
$\%CH_4$	0	0	0	50	50

* Atmospheric air

Experiment 3, was designed to determine the effects of a soil atmosphere containing high concentrations of carbon dioxide on the growth of tomato plants exposed to differing oxygen concentrations in the soil (Table 4). This experiment

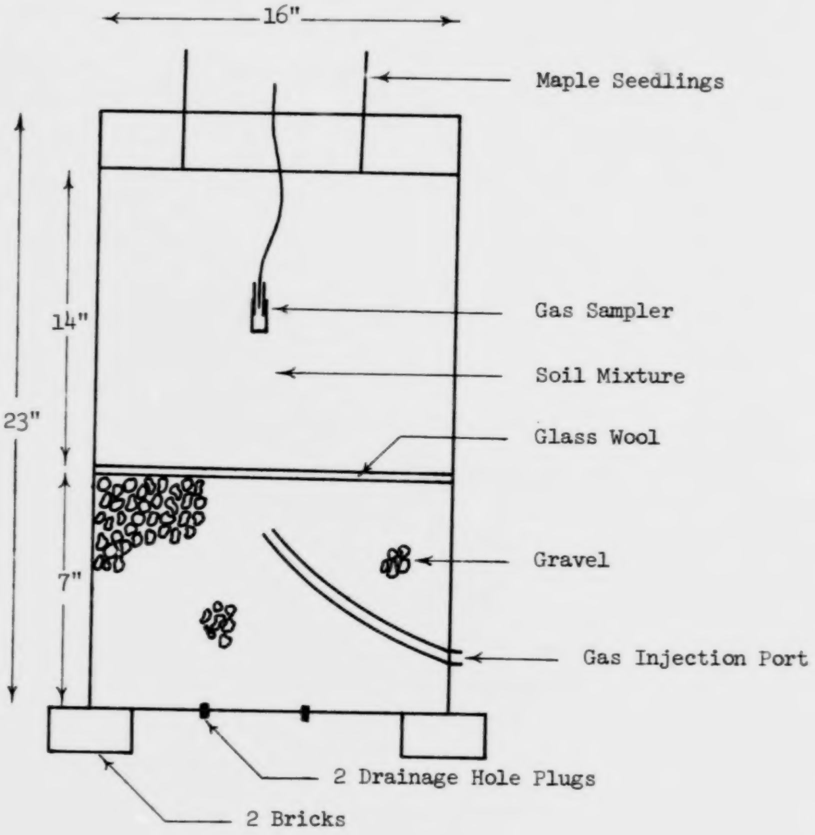
16"
Maple Seedlings
14"
23"
Gas Sampler
Soil Mixture
Glass Wool
7"
Gravel
Gas Injection Port
2 Drainage Hole Plugs
2 Bricks

Figure 2. Modified galvanized steel trash can used to fumigate maple seedlings

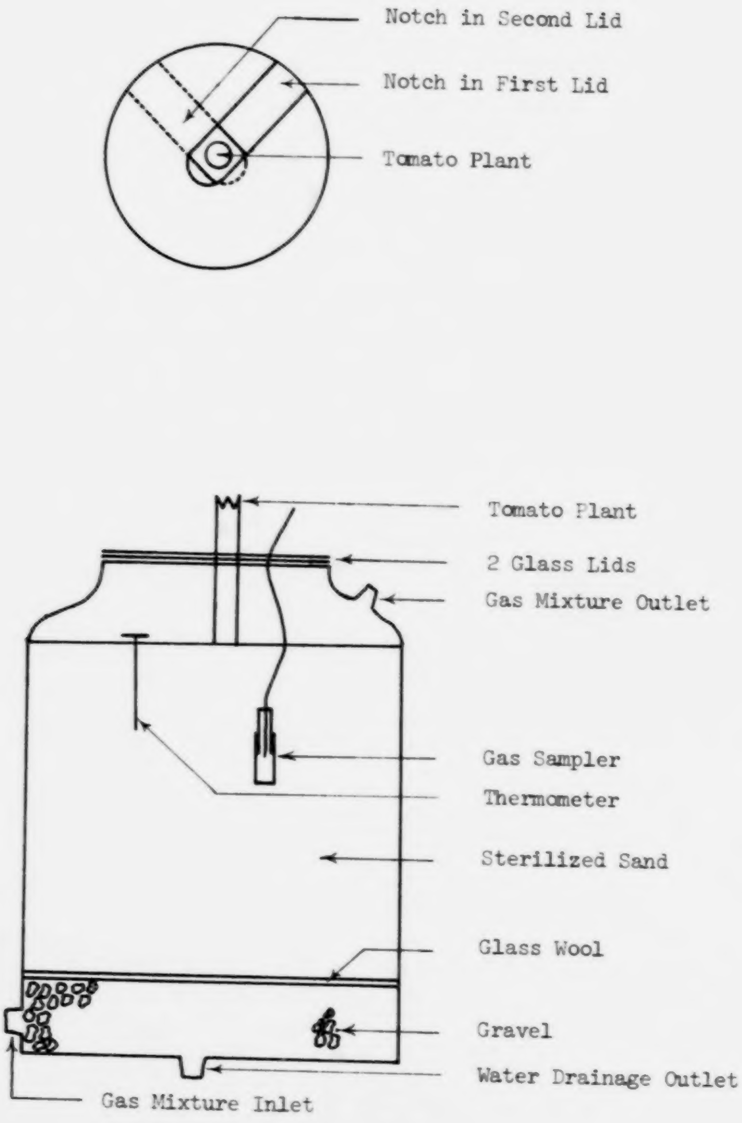
Notch in Second Lid
Notch in First Lid
Tomato Plant
Tomato Plant
2 Glass Lids
Gas Mixture Outlet
Gas Sampler
Thermometer
Sterilized Sand
Glass Wool
Gravel
Water Drainage Outlet
Gas Mixture Inlet

Figure 3. Culture vessel used to fumigate tomato plants

consisted of 12 plants which were divided into 3 treatments with 4 replicates for each.

TABLE 4. COMPOSITION OF ATMOSPHERE USED TO FUMIGATE TOMATO PLANTS IN EXPERIMENT 3

	TREATMENT		
	A	B	C
$\%O_2$	4	15	4
$\%CO_2$	trace	30	30
$\%N_2$	96	55	66

Experiment 4, was designed to determine the effects of a soil atmosphere containing high concentrations of methane or carbon dioxide on the growth of tomato plants (Table 5). This experiment consisted of 16 plants which were divided into 4 treatments with 4 replicates for each.

TABLE 5. COMPOSITION OF ATMOSPHERE USED TO FUMIGATE TOMATO PLANTS IN EXPERIMENT 4

	TREATMENT			
	A	B	C	D
$\%O_2$	5	15	15	5
$\%CO_2$	trace	10	20	trace
$\%N_2$	95	75	65	50
$\%CH_4$	0	0	0	45

The height of the tomato plants was determined by measuring the distance from the glass lid to the tip of the uppermost fully expanded leaf. Adventitious root development was reported only when it occurred above the lid.

The soil atmosphere in the culture vessels was monitored with a Carle model 8500 gas chromatograph as described previously. Total nitrogen content of the dry plant tissue was determined by using the Kjeldahl method, which gives a reading in mg. per 100 mg. of dry plant tissue (31).

RESULTS

Field Studies

Sixty-two trees died by the end of the 1977 growing season, 38 on the experimental plot and 24 on the control. The mean carbon dioxide, methane and temperature were significantly greater (99% C.I.) and the oxygen and moisture content significantly lower on the experimental plot than on the control plot. Bulk density was similar for both plots.

The interpretation of whether a particular species grew significantly better on the control or on the experimental plot depended upon the tree variable measured. On the basis of three or more of these dependent (tree) variables, the majority of species grew significantly better on the control than on the experimental plot. The results of Student's "t" tests for the tree variables (i.e. root biomass, shoot length, leaf weight and basal area) comparing experimental with control plot indicated that black gum exhibited the least variability in growth between the experimental and control plot as indicated by the low Σ"t" value (Table 6). Rhododendron had the poorest growth of all species in that all replicates on both plots succumbed by the end of the winter of 1976-1977, presumably from the abnormally cold temperatures.

Tree data collected in 1976 and 1977 from the five experimental gas barrier techniques and the experimental screening area (which serves as a control for the barrier techniques) for American basswood and Japanese yew are given in Table 7. For American basswood, the gravel/plastic/vents trench and clay barrier mound on the landfill supported significantly better growth (99% c.l.) than did the experimental screening area which had no special treatment and represented typical landfill conditions. This was true for all the dependent (tree) variables measured excluding root biomass. On the other hand, Japanese yew showed no significant differences between the barrier techniques and the experimental screening area.

In order to estimate the relative effect of the soil variables on growth of trees on the landfill plot, multiple, stepwise-regression analysis was performed for American basswood because this species, unlike all the others, was replicated 62 times and, therefore, provided for the best assessment of the effect of the soil factors on tree growth.

Multiple regression analysis of shoot

TABLE 6. RELATIVE TOLERANCE OF SPECIES TO LANDFILL CONDITONS

Rank a	Species	Σ "t" Statistics b
1	black gum	2.66
2	Norway spruce	3.22
3	ginkgo	4.95
4	black pine	6.59
5	bayberry	6.62
6	mixed poplar	8.13
7	white pine	8.94
8	pin oak	8.96
9	Japanese yew	8.98
10	American basswood	9.48
11	American sycamore	10.66
12	red maple	10.95
13	sweet gum	12.62
14	euonymus	14.25
15	green ash	14.87
16	honey locust	15.05
17	hybrid poplar	20.33
18	weeping willow	21.20
19	rhododendron	All plants died

a. Rank 1 - the best growth when experimental plot is compared to the control plot, i.e. most tolerant of landfill conditions.

b. Σ "t" = the sum of the "t" statistics for shoot length in 1976; leaf weight, basal area increase, root biomass and shoot length in 1977 comparing the experimental area with the control.

TABLE 7. MEAN VALUES FOR TREE VARIABLES FOR EACH GAS BARRIER TECHNIQUE FOR 1977

Planting Technique	Species	Root Biomass (mg)	Basal Area (cm^2)	Leaf Weight (g)	Shoot Length 1976 (inches)	Shoot Length 1977 (inches)
Trench-Plastic/,	Jap. Yew	516	17.0	1.24	5.2	2.61
vents/, gravel	Basswood	800	73.3*	3.97*	12.41*	8.98*
Trench-clay/	Jap. Yew	74	9.3	0.41	0.52	2.43
vents ▲	Basswood △	153	0.0	0.02	1.06	.66
Trench-clay	Jap. Yew	616	26.0	0.59	5.1	2.96
no vents	Basswood	1069	23.9	2.21	7.19	5.58
Mounds-no clay	Jap. Yew	1015	14.0	0.64	6.9	1.84
	Basswood	622	31.3	1.89	10.98*	6.66*
Mounds-clay	Jap. Yew	1062	6.2	0.8	7.1	2.78
	Basswood	930	60.0*	3.40*	12.1*	8.52*
Experimental	Jap. Yew	572	45.0	10.98	5.0	2.17
screening area	Basswood	644	26.8	1.04	7.4	3.84

* Grew significantly better than the experimental screening area @ 99% C.L.

△ 4 of the 6 replicates died during 1977

▲ Landfill gas content was significantly higher than all other areas in 1976 length data shows a correlation with the carbon dioxide, oxygen, temperature, bulk density and moisture content (R^2=53%). These same variables explained the differences in leaf weight (R^2=53%), basal area (R^2=63%), and root biomass (R^2=39%).

Simulated Landfill Studies

Both red and sugar maple trees fumigated with the CO_2-CH_4 mixture were in noticeably worse condition than the controls at the termination of the 48-day experiment comparing the effects of simulated landfill gases with those of flooding on two maple species. The main symptoms were chlorosis and abscission of the lower leaves.

The rate of transpiration which is inversely related to stomatal diffusive resistance was significantly less ($P < .01$) for the sugar maples fumigated with CO_2 and CH_4 than the control on day-24. However, red maple seedlings fumigated with CO_2 and CH_4 showed no significant difference in transpiration from the control at any time during the experiment. The sugar maples grown in flooded soil showed a significant decrease in transpiration rate on the 3rd day of the treatment whereas the red maples which were flooded did not show a decrease in transpiration until day-42 of the experiment.

Analysis of variance of the first tomato experiment showed that there was no statistically significant difference between the three treatments (Table 8). All the plants receiving high CO_2 and CH_4 concentrations exhibited adventitious root development on the stems above the glass lids.

TABLE 8. TOTAL INCREASE IN HEIGHT, FOLIAR DRY WEIGHT, AND TOTAL NITROGEN CONTENT OF THE LEAVES OF TOMATO PLANTS* AT THE TERMINATION OF EXPERIMENT 1

	TREATMENT		
	A	B	C
Total Nitrogen (%)	1.9	1.6	1.9
Total Dry Weight (g)	13.1	12.4	13.1
Total Increase In Height (cm)	16.7	10.2	16.7

* Each value is the mean of 7 replicates

In experiment 2 the plants that were treated with air or low O_2, but no CH_4 or CO_2 (Treatments A and B) grew significantly better than did the plants receiving high CO_2 with or without CH_4 (Treatments C and D) (Table 9). The visual appearance of the plants also bore out this relationship.

TABLE 9. TOTAL INCREASE IN HEIGHT, FOLIAR DRY WEIGHT, AND TOTAL NITROGEN CONTENT OF THE LEAVES OF TOMATO PLANTS* EXPERIMENT 2

	TREATMENT				
	A	B	C	D	E
Total Nitrogen (%)	2.58a	2.64a	1.68b	1.89b	1.80b
Total Dry Weight (g)	9.4a	9.1a	3.8c	2.2c	5.9b
Total Increase In Height (cm)	20.9a	21.1a	6.0c	5.0c	16.6b

* Mean of 4 replicates

All values in row followed by an a are greater than values followed by a b or < ($P<0.01$). Values followed by a b are greater than values followed by a < ($P<0.01$).

In the third experiment, the plants that were treated with low O_2 and low CO_2 (Treatment A) grew significantly better than the plants given high CO_2 with high or low O_2 (Treatments B and C) (Table 10). By the 10th day of exposure all the plants given high CO_2 (Treatments B and C) exhibited adventitious root development on the shoots and a general chlorosis of the leaves. The plants remained in this condition throughout the experiment exhibiting little additional growth.

TABLE 10. TOTAL INCREASE IN HEIGHT, FOLIAR DRY WEIGHT AND ADVENTITIOUS ROOT DEVELOPMENT OF TOMATO PLANTS AT THE TERMINATION OF EXPERIMENT 3

	TREATMENT		
	A	B	C
Total Increase In Height (cm)	41.3a	2.4b	0.4b

(continued)

TABLE 10. (continued)

	TREATMENT		
	A	B	C
Mean Dry Weight (g)	9.0a	3.0b	2.5b
Adventitious Root Development	-	+	+

* All values in row followed by an a are greater than values followed by a b. ($P < 0.01$).

In the 4th experiment, the plants that were treated with low O_2 (control) or 10% CO_2 (Treatments A and B) grew significantly better than the plants treated with 20% CO_2 (Treatment C)(Table 11). The latter plants all exhibited adventitious root development and general chlorosis by the 17th day of exposure.

TABLE 11. TOTAL FOLIAR NITROGEN AND DRY WEIGHT AND INCREASE IN HEIGHT AND ADVENTITIOUS ROOT DEVELOPMENT OF TOMATO PLANTS* AT THE TERMINATION OF EXPERIMENT 4

	TREATMENT			
	A	B	C	D
Total Nitrogen (%)	3.1a	2.5a	2.4a	2.0b
Mean Total Dry Weight (g)	11.0a	9.5a	2.0b	8.5a
Total Increase In Height (cm)	41.0a	51.3a	3.8c	29.3b
Adventitious Root Development	-	-	+	+

* Mean of 4 replicates

All values in a row followed by an a are greater than values followed by a b or c ($P<0.01$). Values followed by a b are greater than values followed by a c ($P<0.01$).

By the 17th day of exposure all the plants receiving 45% CH_4 (Treatment D) exhibited adventitious root development. These plants also exhibited chlorosis of the lower leaves.

Discussion

Although landfill gas was not significantly correlated with death of trees in the field screening experiment (r=.238), it was associated with dead trees in a small number of instances. Most of the tree deaths, however, could be attributed to factors other than landfill gas. Low soil moisture, transplanting difficulties, animal damage, and winter injury can explain many of the deaths.

Although landfill gas concentrations in the experimental plot were not high enough to account for actual death of many of the plants, representative landfill soil conditions were of adequate magnitude to detect the order of relative tolerance of the surviving trees as listed in Table 6. This listing resulted from a consideration of four tree variables including leaf and root biomass, shoot length (1976 and 1977), and basal stem area.

It is interesting that of the nine most tolerant species, only three i.e. black gum, bayberry and pin oak, (26) have been reported to be able to withstand low oxygen tension in the soil, one of the criteria for selecting experimental species (Table 1). However, seven of the first ten species were 3-feet or less in height when planted, whereas seven of the last ten species in Table 2 were 6-feet or taller when planted. Possibly the size of the tree as well as the biological ability of species to withstand low oxygen, is important in selecting vegetation for completed sanitary landfills.

In order to assess the role of the selected soil variables in predisposing these species to landfill tolerance, multiple regression analysis was performed. Results indicated that the soil variables oxygen, carbon dioxide, temperature, moisture content, and bulk density explained a significant portion (95% C.I.) of the discrepancies in tree responses to landfill conditions.

The effectiveness of each gas barrier technique in preventing methane gas migration into the trenches may be evaluated by considering the ratio between the methane concentrations around the periphery of the trench and those inside the trench. In the gravel/plastic/vents trench and clay barrier trench, the ratios are 207:1 and 54:1

respectively, indicating that these trenches have functioned effectively in keeping out methane gas. On the other hand, for the clay/vents trench, the 1:1 ratio indicated that this gas barrier technique was not effective in preventing the migration of methane from the refuse into the trench. Landfill gas was never detected in either of the two mounds. Vegetation growth was also best in the gravel/plastic/vents trench and the two mounds.

Sugar maple was intolerant of flooding as evidenced by the statistically significant decrease in transpiration rate after only one day of flooding and the loss of all leaves by the termination of the experiment. The transpiration rate of red maples did not decrease until the 42nd day of flooding. The fact that red maple is more tolerant of flooding than sugar maple has been reported in the literature (21).

Flood tolerance has been attributed to more than one adaptive mechanism in several species (25). The adventitious root development on the flooded red maples in this experiment have been found to occur on many other "flood tolerant" species and are believed to contribute to flood tolerance to some degree. Such an adaptation was not observed on the red maples fumigated with simulated sanitary landfill gas mixtures. This is not surprising since adventitious root development is dependent upon the presence of water. Other adaptations which are believed to contribute to flood tolerance which are not as water dependent include the ability to withstand elevated levels of CO_2 in the soil (25) and to undergo anaerobic root respiration without the production of inhibitory concentrations of ethanol. Mechanisms such as these could explain why the differences between the two species were more pronounced in their response to flooding than to soil contamination with simulated landfill gas. The inability to develop adventitious roots in response to landfill gas contamination would reduce the advantage enjoyed by flood tolerant species whereas other mechanisms contributing to flood tolerance such as the ability to withstand elevated levels of CO_2 in the soil or undergo anaerobic respiration in the roots which are not inhibited by lack of water and would continue to supply some protection.

Carbon dioxide contamination in the root zone of tomato plants in solution culture was toxic when the concentration of CO_2 averaged 17.0% during the experimental period. When CO_2 concentrations averaged 8.8% or less no symptoms were observed, indicating that there was a threshold level between 9 and 17% at which CO_2 became toxic under these experimental conditons. Tomato plants exposed to 17% CO_2 exhibited progressive chlorosis and abscission of the lower leaves, adventitious root development, chlorosis of the entire plant, and a reduction in the growth rate. Exposing tomato roots to concentrations of CO_2 between 25 and 36% resulted in earlier and more severe sympton development on tomato plants than did 17% CO_2.

These findings are consistent with what is reported in the literature. Erickson (11) in 1946 found that 28% CO_2 in the root zone severely reduced the growth rate of tomato plants.

No interaction was observed among O_2, CO_2 and CH_4 when they occurred together in the root zone in terms of symptom development on tomato plants.

CONCLUSIONS

1. Black gum was the species most tolerant to field conditions at the field landfill plot.
2. Honey locust made the poorest growth of all surviving species on the field landfill plot and appeared to be least adapted to landfill conditions.
3. Soil carbon dioxide, oxygen, moisture content, bulk density and temperature were important soil factors controlling the growth of American basswood on the experimental plot on the Edgeboro sanitary landfill.
4. Soil mounds, either with or without an underlying clay gas-barrier, functioned successfully in preventing the migration of landfill gases into the root zone of trees. A 4 foot deep trench with a 1 foot layer of road gravel overlain with polyethylene sheeting and vented with perforated vertical vent pipes also functioned to keep out the gases of anaerobic decomposition.
5. Woody plants appeared able to better survive on a completed sanitary landfill if planted when small in height i.e.

less than 3 feet. During this study, this factor appeared to be more important than the biological ability of a plant to withstand low oxygen environments.

6. Red maple, which is flood tolerant, was found to be more tolerant of soil contaminated by simulated landfill gas than sugar maple, which is not tolerant of flooding.

7. Tomato plants growing in sand-solution greenhouse cultures were severely damaged by exposure to carbon dioxide concentrations of 17% or greater in the root zone. This response was not influenced by the presence or absence of high concentrations of methane or fluctuations in the O_2 concentrations, provided the O_2 in the root zone was not less than 2%.

RECOMMENDATIONS

1. Those responsible for planting vegetation on completed landfills should avail themselves of current research on the adaptability of species to landfill conditions and avoid the use of non-tolerant species.

2. The use of a barrier technique for excluding landfill gas from the root zone of vegetation should be considered when planting on a former landfill. Two methods to consider are: (a) a mound of soil over the existing cover, (b) a lined and vented trench backfilled with suitable soil.

3. The use of smaller planting stock might also increase the chance of survivability.

4. Adequate irrigation of the plants established on a landfill is an important contribution to their survivability.

5. Further studies should be undertaken to determine if the ability to withstand high levels of carbon dioxide in the root zone is a characteristic of flood tolerant species.

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EFFECTS OF PARTICLE SIZE ON LANDFILLED SOLID WASTE: COLD CLIMATE STUDIES

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ABSTRACT

This research program was initiated with the overall objective of determining the effects of milled refuse particle size on landfilled solid waste.

Four experimental test plots were constructed using four different particle size distributions. Each pit representing a specific milled refuse particle size will be evaluated in an attempt to correlate particle size with different parameters. The parameters under study are: wind displacement of milled refuse, differential settlement of compacted milled refuse, and attraction of vectors, rodents, birds and wildlife to the milled refuse.

Monitoring of the test pits was just initiated so preliminary results are not available to date. Particle size distribution curves are developed herein for the milled refuse used. Four breakdowns of the composition of the raw solid waste prior to milling are included. Moisture content and field density test results are shown herein.

This report was submitted in fulfillment of Grant R805012 by the Montana State Department of Health and Environmental Sciences under the sponsorship of the U. S. Environmental Protection Agency. This report covers the period from July 25, 1978, to December 20, 1978, and work will be completed on July 3, 1980.

INTRODUCTION

BASIS FOR STUDY

Design criteria is inadequate or totally unavailable, to date, for correlating milled refuse particle size with wind displacement problems, settlement differentials, vector problems, etc. This lack of design criteria or operational parameters could have a detrimental effect on future milled refuse operations. States need more detailed data to establish acceptable parameters for operation and maintenance of shredding facilities. They also need more detailed data to adequately adopt proper regulations governing the disposal of milled refuse. Some states are adopting regulations that define particle size limitations without any proven data to substantiate these limitations. Reliable design criteria is needed to relate the effects of particle size on operational and esthetic variables.

PARAMETERS TO BE EVALUATED

The purpose of the project is to evaluate the following variables:

1. The effect of wind velocities on the displacement of compacted milled refuse without cover.
2. The amount of differential settlement occurring in compacted milled refuse correlated with particle size.
3. The initial density in compacted milled refuse and subsequent density with relation to time resulting from consoli-

dation of refuse using varying particle sizes.

4. The presence or absence of vectors, rodents, birds, and wildlife correlated to particle size of refuse.

5. Any observations of additional parameters that can be correlated with the different particle size will be noted throughout the project period.

PARTICIPANTS IN STUDY

This project is being funded by a research grant through the Solid and Hazardous Waste Research Division of the Municipal Environmental Research Laboratory of the Environmental Protection Agency and by the Solid Waste Division of the Montana State Department of Health and Environmental Sciences.

This project was undertaken by the Montana Department of Health and Environmental Sciences in cooperation with the City of Great Falls. The firm of Thomas, Dean & Hoskins, Inc., of Great Falls, Montana, was utilized to provide the consulting engineering services required. The City of Great Falls refuse shredder was used to produce the different particle sizes of refuse for placement in test pits.

EXPERIMENTAL WORK

PROCEDURAL REQUIREMENTS FOR STUDY

The Environmental Protection Agency was very specific on the methods and procedures they wanted followed to develop the research data needed. Project work was not initiated until the Environmental Protection Agency, Montana State Department of Health & Environmental Sciences, and Thomas, Dean & Hoskins, Inc., were all in agreement as to the methods and procedures that would be followed to complete the research work.

In general, the project was to include four test plots or holes with dimensions 20 feet wide, by 35 feet long, by 6 feet deep excavated in the ground. Each test plot was to be filled with compacted refuse of a specific particle size. The milled refuse in the first test pit was proposed to ideally have 70 percent passing a two inch particle size, the second pit, 70 percent passing a four inch particle size, the third, 70 percent passing a six inch particle size, and the fourth pit, 70 percent passing an eight inch particle size. The test plots were to be located end to end with the long side of the four pits facing the prevailing southwest winds. These test pits were to be located in such a configuration and at a location that would be unobstructed in regard to the prevailing winds.

Following completion of the installation of the refuse in the pits, each pit was to be enclosed with one-half inch mesh fencing.

Monitoring data over a two year period is to provide information on wind displacement and settlement of the differing sizes of milled refuse. It also is to provide a correlation between vectors, rodents, birds and wildlife attraction to the varying particle sizes.

FIELD CONSTRUCTION OF TEST PITS

On August 18, 1978, Thomas, Dean & Hoskins, Inc., was given the go-ahead to initiate the project work required for the study.

The site selected for construction of the test pits was located on a hill one-quarter mile east of the existing City of Great Falls landfill. This site was selected because the wind blows across the site unobstructed by any physical features such as trees, buildings, etc. The haul distance to the test site was approximately equal to that required to get to the City landfill site.

Four test pits were excavated with dimensions of 21'x36'x6' deep. A Caterpillar Traxcavator Model 977 with a 2-3/4 cubic yard bucket was used to excavate and shape the four test pits. Each test pit was separated from the adjacent pit by 8 feet of undisturbed soil. Access to the interior of each pit was down a steep slope into the hole.

Following completion of the excavation phase of the test pits, a carpenter installed the plywood sidewalls. These plywood sidewalls were installed to provide a pit with the exact dimensions of 20 feet wide by 30 feet long. It is impossible to excavate a hole with exact dimensional tolerances so the plywood sidewalls were installed to assure that the pit dimensions would be accurate. Knowing the exact dimensions of the pit, it would be possible to determine the volume and subsequently the wet density of the "in-place" refuse. The plywood sidewalls were installed to a height six inches below the existing ground surface to keep from obstructing any wind from blowing against the top surface of the compacted refuse. Two by four framing provided support for the plywood sidewalls. An 8 foot long portion of the plywood sidewall construction was not installed until after the compaction equipment was driven into each test pit.

The milled refuse for the test pits was provided by the City of Great Falls shredding facility. This solid waste reduction plant was constructed in 1975 with two Heil vertical shaft grinding mills; one 20 ton per hour unit and one 15 ton per hour unit.

The shredding facility has a magnetic separator unit for removal of ferrous metals installed on the outlet conveyor belt. At the time that this project was initiated, the magnetic separator was not in operation. Therefore, the milled ferrous metal material was included in the refuse placed in the test pits.

Weigh scales are installed at the waste reduction plant to weigh all incoming refuse and all milled refuse prior to disposal. All the refuse delivered to the test pits was weighed on these existing scales.

It was proposed that the mills would produce a specific, but different, particle size distribution of refuse for installing in each of the four test pits. We constructed a sieve unit or trommel for analyzing a milled refuse sample in regard to particle sizes. The trommel was designed to sieve out particle sizes of dimensions 1/2 inch, 1 inch, 2 inch, 4 inch, 6 inch, and 8 inch. Hammer adjustments in the mills could then be made to approximate the size of particle required.

The sieve unit consisted of two 13.5 ft. long circular trommel units, 22 inches in diameter with a support frame. Each of the two cylindrical units contained three different screens or sieve sizes. Two trommel units were required to process the six different mesh sizes required. They were designed to be manually turned by hand. The upper end of the trommel frame was raised above the floor 13-1/4 inches to allow particle flow from one end to the other. The refuse would be placed in the end of the trommel unit with the smallest sieve size near the head end of the unit. As the trommel was turned, the refuse would fall and tumble through the one-half inch screened area down to the second screen with one inch openings, past the third screen with two inch openings, and out the end of the trommel unit onto a container on the floor. At this point, the refuse would be rerun through the second trommel unit sieving it to 4 inch, 6 inch, and 8 inch particle sizes. Anything larger than 8 inch would pass out the end of the trommel. In this way, it was possible to run a sieve analysis using six different sieve sizes.

Grinding shredded refuse to a consistent particle size was an area of concern during the placement operation. Hammer mills such as Great Falls' unit tend to be subject to hammer wear on a day-to-day basis. As the hammers wear, the particle size of the material varies also. The more the hammers wear, the larger becomes the particle size. So it was important that once shredding was started and placement begun, the operation continued uninterrupted until a specific pit was complete. Getting the refuse to the site, into the pits, and compacted in place without delays was difficult due to the many operations that had to be coordinated at one time.

Milled refuse was hauled to the pit in 65 cubic yard transfer trailers which carry a pay load of approximately 30,000 to 32,000 pounds per load. Five to six loads were required per pit. The pulverized material was compacted into the 65 cubic yard transfer trailer at the pulverizer facility. The compactor unit that presses this material into the truck does so obtaining a wet density of approximately 500 lbs. per cubic yard.

For each 65 cubic yard load of refuse

milled for placement in a test pit, one composite sample was sieved and one moisture content was determined. To get the samples for these tests a 30 gallon container was placed underneath the bottom of the discharge conveyor belt as the refuse tumbled off the conveyor belt prior to compaction into the transfer trailer. The 30 gallon container was partially filled each time a sample was taken to develop a composite sample for each 65 cubic yard load. The moisture samples were collected in plastic bags and dried in the ovens at our laboratory following the days run.

When the 65 cubic yard transfer trailer was completely filled, it was then weighed and sent to the test pit site area. The refuse was then stockpiled at the site. A John Deere backhoe-loader was used to pick up the refuse and place it in lifts into the specified pit. Once the refuse was discharged into the pit, the backhoe was used to spread the refuse to a uniform depth of 18" to 24". Then a Caterpillar dozer on tracks located down in the bottom of the pit was used to drive over the material and compact it to the required density of approximately 1,000 pounds per cubic yard.

Placement of the refuse depended on many different operations. The City pulverizer experienced some breakdowns which stopped production of the milled refuse and subsequently shut down the placement of the refuse in the pits. Also, there were some backhoe and dozer equipment breakdowns at the test pit site which shut down the refuse placement operation. Excessive windy weather, rain, and snow conditions sometimes shut down the placement operation also.

Field installation of the milled refuse required two pieces of equipment. A John Deere backhoe was used to spread the milled refuse in lifts. Then a Caterpillar Traxcavator which we located down in the pit was used to compact the refuse. Compaction was essentially complete when there was no indentation left by the dozer tracks on the surface of the refuse. This required about six passes over the refuse with one track.

The Caterpillar Traxcavator weighed 36,500 pounds with a track width of one foot six inches and an on-ground length of nine feet six inches. The three-bar street pads on the tracks provided a compressive downward force of 1,281 pounds per square foot.

Following completion of the installation of each load of approximately 65 cubic yards of refuse, measurements were made to determine the in-place wet density of the load. Knowing the weight of the delivered load of refuse and the field measured volume of the compacted refuse in place, the wet density could be determined. See Figure A-1 for the location of the density check points. We attempted to maintain a wet density of approximately 1000 lbs. per cubic yard in place. Densities varied from approximately 835 up to over 1,364 lbs. per cubic yard for an average somewhere around 1,000 lbs. per cubic yard for all pits. It was our opinion that obtaining a density of 1,000 lbs. per cubic yard was probably in excess of what most landfill sites accomplish in Montana. Heavier densities may be obtained in areas where land is scarce, but in Montana where land is more readily available there is generally less compactive effort applied to the waste.

To measure settlement that occurs within the lower portions of the pit, we installed two foot diameter steel discs approximately 4 to 5 feet above the bottom of each pit. The reason for installing the plates 2 to 3 feet below the surface was that the refuse in the top portion of each pit will probably be displaced by wind velocities throughout the two year study period. With these plates installed below the surface, we can determine what settlement takes place in the different pits throughout the period. The 1/4 inch metal discs were installed, four per pit, at the locations given in Figure A-1.

Following completion of placement of the refuse in each pit, we excavated back down to the surface of these metal discs and installed 4 inch diameter pipe. This was to allow for access to the top of the metal discs throughout the project period. We attempted to dig through the compacted refuse by hand which proved almost impossible. But we were finally able to use a gasoline powered fence post auger and very carefully dig down through the refuse to the metal discs. Then we cleaned the holes out by hand and installed the 4 inch diameter pipes for access to the tops of the metal plates.

Following completion of the placement of the refuse and the installation of these plates, the pits were covered with plastic to prevent the refuse from blowing and being displaced prior to installation of the fenced enclosure.

The fencing encloses four sides of each pit allowing for a three foot buffer area on the sides of each pit and a four foot buffer area on the ends of each pit. The refuse that is displaced by the wind will settle in this buffer area where it can be collected for weight determination during the monitoring stage of the data collection phase of this project. The fencing material around the periphery of each pit is 1/2 inch mesh to allow for retaining the 1/2 inch or larger material within the pit area. Anything smaller than 1/2 inch may get through the fence and be blown away.

Originally, the area over the top of each test pit was to be enclosed with a chicken wire mesh to prevent the refuse from soaring up out of the pit. However, the Environmental Protection Agency felt that this top covering should be eliminated to allow for bird access to the area, since this is one of the items to be noted in the project.

DATA COLLECTION AND FIELD MONITORING

Previous mention has been made of the sieve tests that were conducted during the placement of the refuse material in the test pits. Also, moisture tests and inplace density tests were taken during that same period of refuse placement. In addition to the test data recorded on the initial construction phase of the project, additional data will be collected throughout the remaining project period. As mentioned before, settlement determinations are to be made on the test plates installed on the interior of each test pit. Also, elevation checks are to be made on the test pit surfaces, Figure A-1, on a periodic basis to determine if settlement takes place on the surface of the pit. However, it should be noted here that any displacement of the material, due to wind, will also lower the top working surface of the pit and, therefore, it will be difficult to determine whether settlement took place on the surface or whether the material was actually displaced due to wind. However, as time progresses and the top surface builds up a scum type layer preventing wind displacement of the material, then it will be possible to determine if settlement takes place due to deterioration of the pit itself. Other measurements in regard to vector, rodent activity, and bird activity, are to be made.

A comparison shall be made to determine the number of flies attracted to each plot to see if the different particle size can be correlated with fly activity. A Scudder Grille shall be used to attract the flies for counting. Fly counts shall be made once per week, from May through September the first year, and once per month from May through June 30, the second year.

Rodent activity shall be checked in an attempt to correlate it with the size of the milled refuse. Since each test plot is to be fenced, two small 6 inch diameter access holes were cut in the fencing on the northeast and southwest sides of each test plot. This will allow the rodents to gain access to the area should they desire. Rats have never been observed in the Great Falls area, but gophers and field mice are common. To ascertain the rodent species, snap traps shall be set in the area of any runs that develop. Trap checks shall be made once per week from May through September the first year, and once per month, from May through June 30, the second year.

Bird activity shall be monitored for each test pit. During each monthly or bimonthly visit to the site, bird activity shall be noted in and around each test plot. The test pit is open on top to allow for bird access to the refuse.

Analysis of wind effects on the compacted refuse will continue throughout the project period. Measurements of wind correlated with amount of refuse displaced is critical in the early stages of the project following material placement in the test pits. Wind direction and velocity data shall be provided by the U.S. Weather Bureau for the periods of collection to see if any correlation exists between the particle size of the material displaced and the average and maximum wind velocity data for that period. Temperature and precipitation data will also be collected.

RESULTS TO DATE

At the time of this writing (December, 1978) results are limited. Tests performed during construction of the test pits are complete, but field monitoring following installation of the pits is just beginning. The only parameter monitored during this initial two week period has been wind data. When the oral presentation is made, more data should be available.

Located in the appendix of this report are curves indicating the different results of the sieve analysis for each pit. Figure A-2 through A-5 indicates the variation in the particle sizes within each pit. There is a considerable range of variation due to hammer wear in the mills themselves over the period while the refuse was being placed in the pits. It usually took from two to three days to get one pit filled with compacted refuse.

We attempted to obtain a uniform particle size for each pit but found out that this was impossible due to limitations of the hammers in the mill and also the limitations of the raw domestic refuse prior to milling.

In this project, we were not required to run sieve analyses on the raw refuse, although we did run one sample through just for informational purposes. While attempting to mill an 8 inch particle size for the last pit, we found it impossible to obtain 70 percent passing (30 percent retained) an 8 inch screen. Because of this difficulty, we wanted to determine whether this limitation was caused by the configuration of the hammers inside the mills or by the size of the incoming raw refuse. As a result, we ran the test on the raw refuse and found that the raw refuse itself without even milling would not meet the criteria of 30 percent retained on an 8 inch screen. The one sample we did run (Figure A-6 in appendix) on the raw refuse indicated that 30 percent of the sample was larger (70 percent passing) than a 4-1/4 inch screen. With a normal gradation it would then be impossible to get anything larger than what the raw refuse gradation was coming into the pulverizer unit. With the majority of the hammers removed from the mill, the largest size particle we could mill was used for pit #4. A 6 inch particle size (70 percent passing) was used for the top lift of pit #4 and this was the largest particle size of any loads delivered to the test site.

The curves for pit No. 1 indicate that the top lift has 70 percent passing a 2-1/2 inch sieve. However, the lower lifts in pit No. 1, lifts one through four, vary in size from approximately 3-1/2 inches to 4-1/2 inches at the 70 percent passing line. Pit No. 2 has a top lift at approximately 4-1/2 inches particle size with the lower lifts varying from 4-1/2 inches down to 3 inches in particle size at the 70 percent passing line. Pit No. 3 also has a top lift of approximately 4-2/3 inch particle size with the lower lifts varying from 4-1/2 inches up to 5-1/2 inches approximately at the 70 percent passing line. Pit No. 4 has a top lift using a 6 inch particle size. The lower lifts in pit No. 4 varied from approximately 4-1/2 inches up to 5 or 5-1/4 inches in diameter at the 70 percent passing line. This was the largest particle size in any of the four pits.

On November 4, wind gusts in excess of 80 mph fanned a fire at the City landfill site and wind-carried cinders started a fire in test pit No. 3. The fire ruined the pit and chances to obtain any future test data from it. Pit No. 3 is therefore eliminated from the project work.

The composition of the raw solid waste as sampled for each test pit is shown in Table A-1 in the Appendix. The category for "Ash, Rocks & Dirt" was included under fines since the material was too small to separate from other fines. Grass cuttings were also included under fines except for in pit No. 1 where an attempt was made to place it under garden waste. The average composition in percent by wet weight shown represents the typical composition of municipal waste for the Great Falls area as would be received in the fall of the year.

The table also indicates for comparison purposes only, data from the Proceedings of the 1970 National Incinerator Conference and the Land Disposal Project for Boone County Field Site, Walton, Kentucky. With the data modified to include the same parameters, the Great Falls refuse falls near or within the range of the other two sets of data. Our metal content was slightly higher than the other studies revealed but this was due to one sample being high. We feel this is a general indication that the refuse used in the test pits was similar to that produced by other areas.

Density and moisture content data is shown in Table A-2 in the Appendix. An attempt was made to try and be consistent in the compactive effort applied to each lift of refuse but this was not always done. The densities of the lifts varied from 835 lbs./C.Y. up to 1,364 lbs./C.Y. with overall densities of 1,117; 991; and 987 lbs./C.Y. for pits 1, 2 and 4 respectively. This compares closely to our goal of 1,000 lbs./C.Y. Pit No. 3 was omitted from the data determination due to the fire which destroyed the pit as previously mentioned.

The dry weight moisture contents varied from 13.2 to 32.01 percent for the different lifts with the averages being 19.62, 20.85 and 21.19 percent for pits 1, 2 and 4 respectively. Normal moisture contents for municipal refuse vary from approximately 20 percent up to 60 percent so these percentiles in Great Falls would be on the low side of this range.

On November 29, 1978, the test pits were completed with the fencing installed and the plastic covering removed. On December 1, 1978, a snow storm covered the test pits with a snow cover that increased with later storms. Windy periods caused a heavy crust to develop on the snow and as of this writing on December 20, 1978, (a written copy of each presentation had to be submitted by January 1, 1979), the surfaces of the test pits were still unchanged. A warm-thaw with prolonged temperatures above freezing without additional precipitation will be required to dry the surface of the pits. Winds in excess of 40 mph have recently occurred although little refuse has been displaced due to the snow crust on the pits. Internal settlement and surface settlement of the test pits will not be evaluated for the first month's operation until the end of December. Fly counts and rodent trapping are not checked during the winter months and there has not been any evidence of bird activity at the site as of this writing.

To date it is too early to recommend any specific size of milled refuse particle as having advantages over a smaller or larger size. Field monitoring has just begun and will continue for another 1-1/2 years providing more data as time progresses.

If one particle size is proven to have advantages over any other particle size, it will be difficult to consistently mill the refuse to the preferred particle size with the vertical shaft mills installed in Great Falls. The sieve analyses indicate that when grinding the smaller particle sizes, the Heil units will not consistently produce one size particle. The continual increase in particle size is caused by the normal wear of the hammers. When milling the larger particle sizes of refuse, the particle size varies with the particle size distribution of the incoming raw waste. Hammer wear would be experienced on any brand of shredder unit and is not unique to just the Great Falls Milling units. It is possible that a change in the design of some grinding mills may be required if a limitation is to be placed on the size of particle that must be produced.

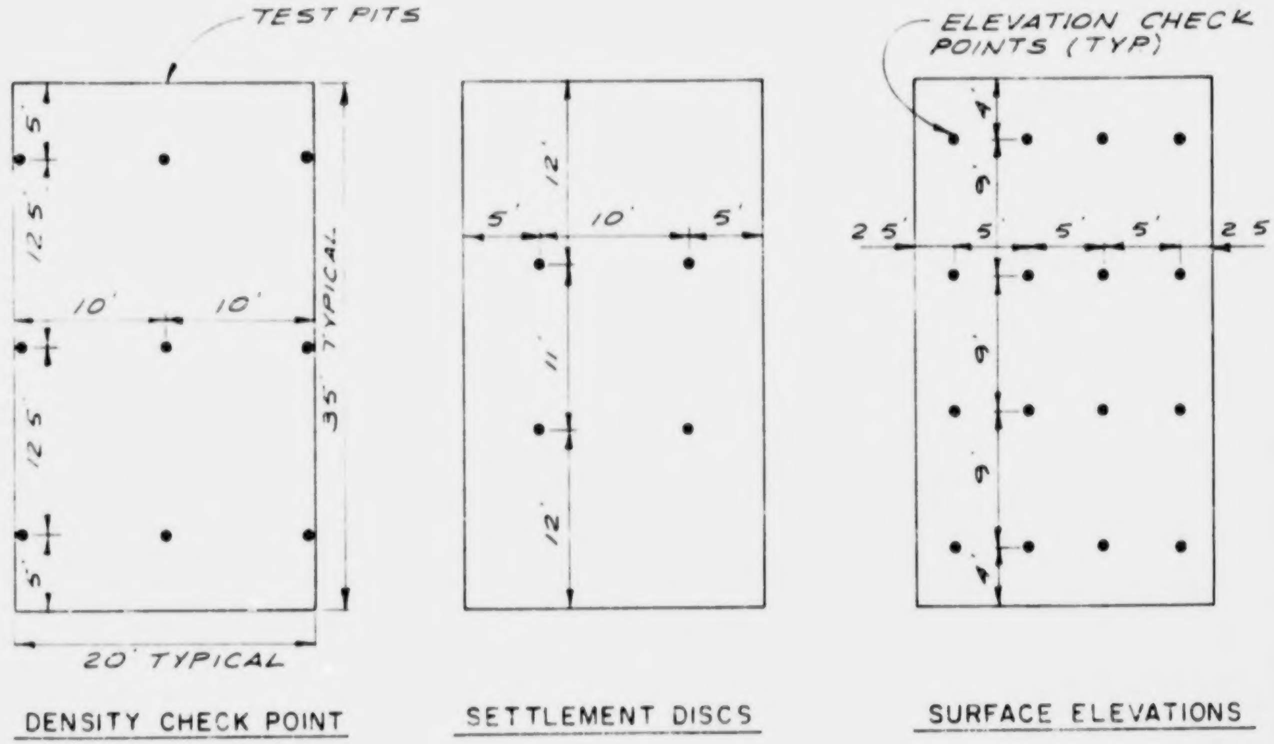
TEST PITS
5'
12.5'
12.5'
5'
10'
10'
35' TYPICAL
20' TYPICAL
DENSITY CHECK POINT
5'
10'
5'
12'
11'
12'
SETTLEMENT DISCS
ELEVATION CHECK POINTS (TYP)
4'
9'
9'
9'
4'
2.5'
5'
5'
5'
2.5'
SURFACE ELEVATIONS

FIGURE A-1 ELEVATION CHECK POINTS

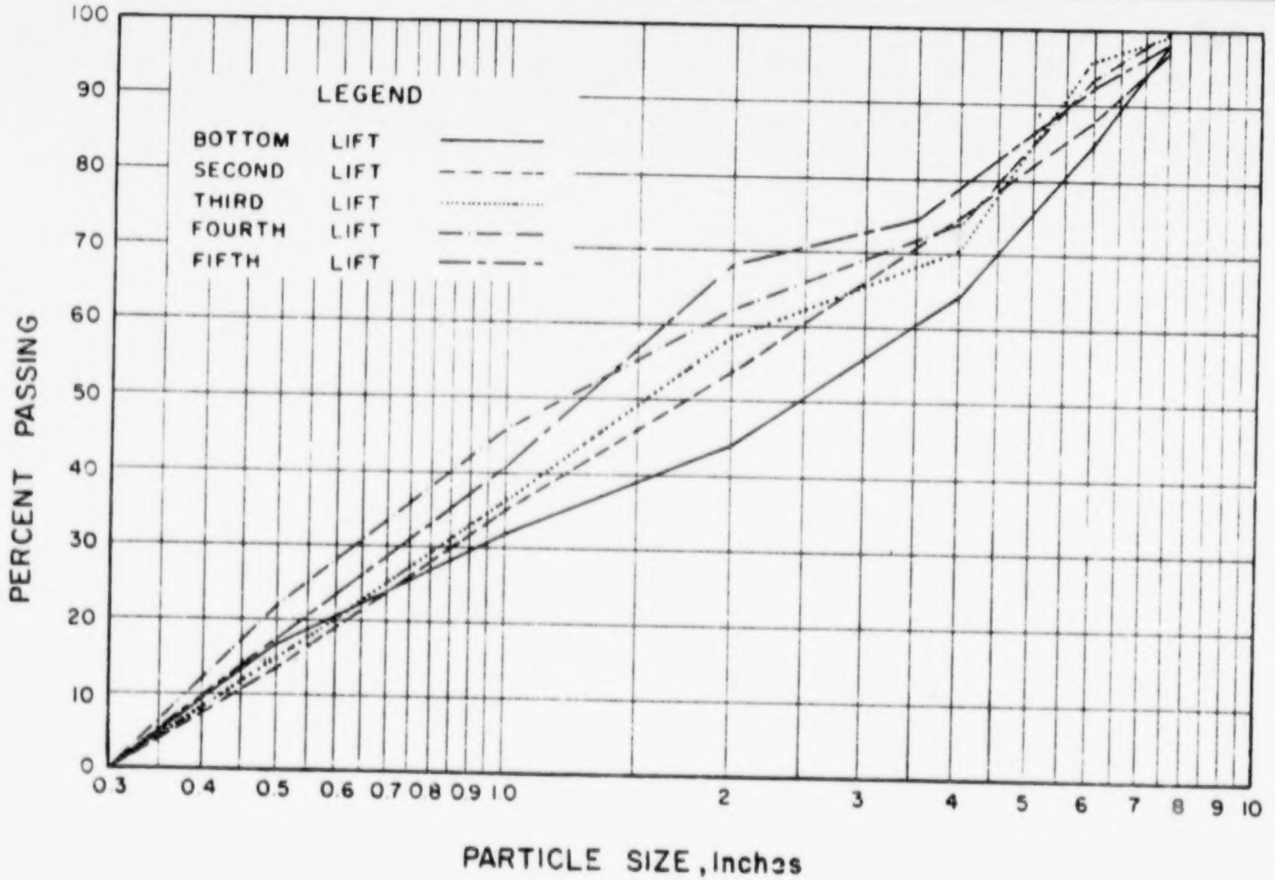

Figure A-2 PARTICLE SIZE DISTRIBUTION CURVE - TEST PIT #1 - MILLED REFUSE

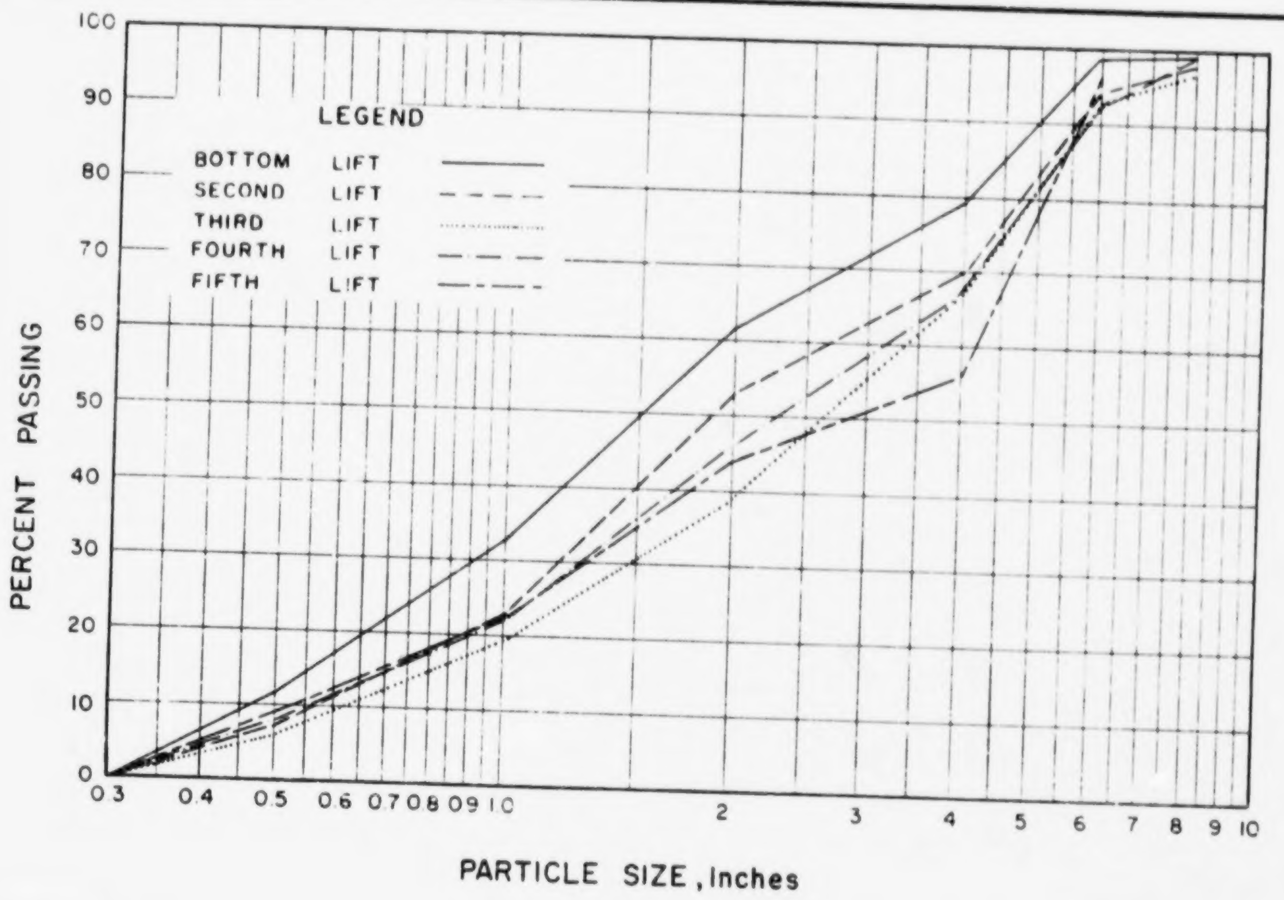

Figure A-3 PARTICLE SIZE DISTRIBUTION CURVE TEST PIT #2 - MILLED REFUSE

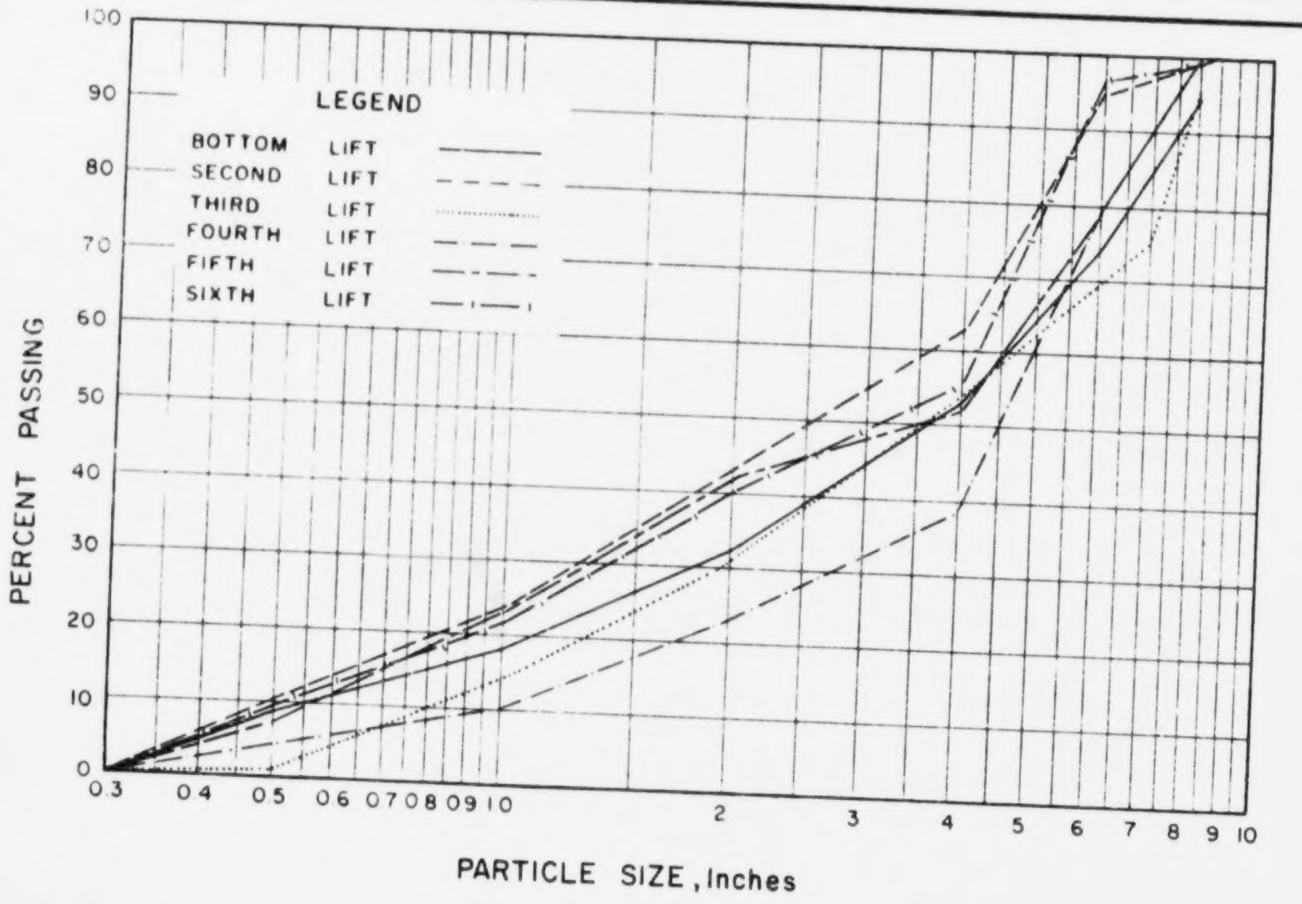

Figure A-4 PARTICLE SIZE DISTRIBUTION CURVE - TEST PIT #3 - MILLED REFUSE

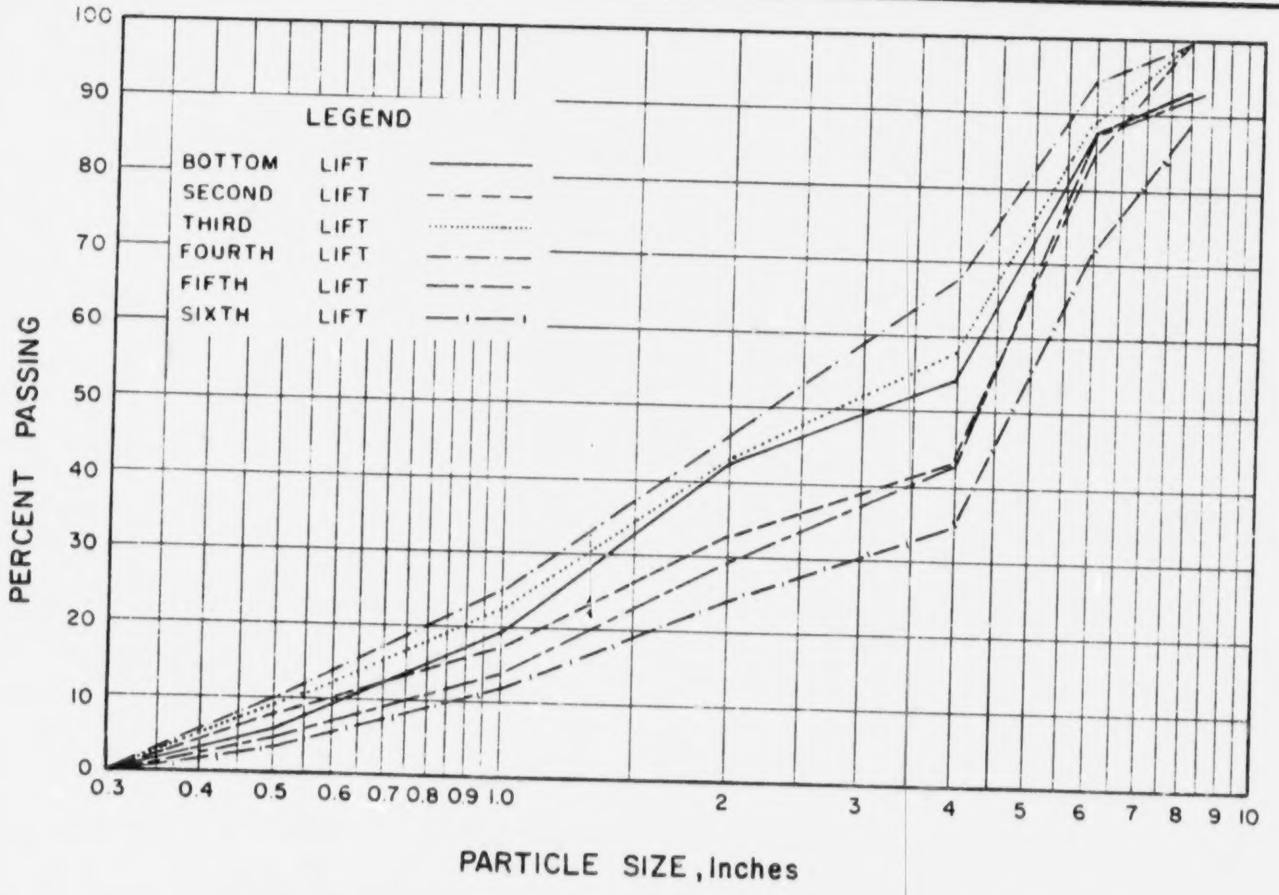

Figure A-5 PARTICLE SIZE DISTRIBUTION CURVE - TEST PIT #4 - MILLED REFUSE

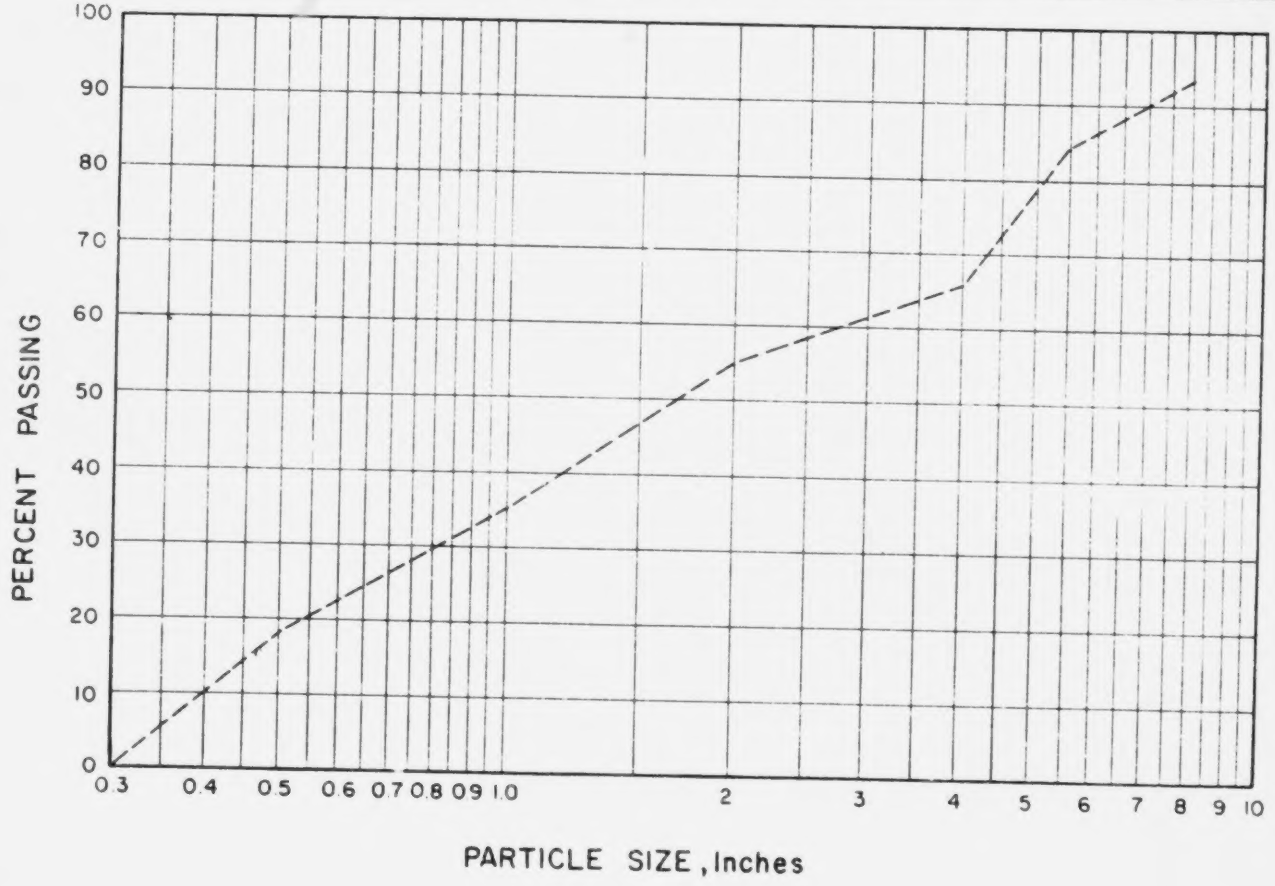

Figure A-6 PARTICLE SIZE DISTRIBUTION CURVE - RAW WASTE SAMPLE

TABLE A-1 REFUSE COMPOSITION COMPARISIONS

Sample Location	Total Weight Separated (pounds)	COMPONENT-PERCENT BY WET WEIGHT								
		Food Waste	Garden Waste	Paper	Rubber, leather, and Textiles	Wood	Metal	Glass	Plastic	Ash, rock, dirt, fines and Miscellaneous
Pit #1	98.4	8.1	△10.4	55.8	9.6	0.2	5.4	6.8	2.7	1.0
Pit #2	136.9	9.4	1.9	38.5	2.2	3.6	16.9	9.9	8.4	9.2
Pit #3	219.7	1.9	22.7	28.1	0.4	0.9	7.9	4.9	5.5	27.7
Pit #4	102.9	5.0	9.7	35.8	2.2	1.2	13.2	5.7	7.3	19.9
Avg. of all pits	---	6.1	11.2	39.6	3.6	1.5	10.8	6.8	6.0	14.4
Avg. of Pits *		8.2	Excluded	53.3	4.8	2.0	14.5	9.1	8.1	Excluded
1968 Annual Avg. Ⓧ		19.3	---	51.6	4.6	3.0	10.2	9.9	1.4	---
Boone Cnty. Kentucky +		4.5	9.2	49.2	10.3 ◇	4.1	9.2	7.1	Incl. w/ rubber	6.3
Boone Cnty. w/o Misc. S		5.4	Excluded	58.3	12.2 ◇	4.9	10.9	8.4	Incl. w/ rubber	Excluded

△ Grass included here for this test only.

* Excluding garden waste, ash, rocks, dirt, fines and miscellaneous.

Ⓧ Estimated annual average National Refuse Composition on a yard waste-free & misc.-free basis-1970 Natl. Incinerator Conference Proceedings.

+ Land Disposal Project Boone County Field Site, Walton, Kentucky, Jan. 1973.

◇ Includes plastics.

S Excludes ash, rock, dirt, fines & misc. garden waste.

TABLE A-2 DENSITY-MOISTURE SUMMARY SHEET

SAMPLE LOCATION	LIFT	DENSITY ⓧ LBS/C.Y.	OVERALL* DENSITY LBS/C.Y.	MOISTURE % BY DRY WEIGHT
Pit #1	1	879		14.76
	2	1099		19.21
	3	1104		31.23
	4	1128		17.93
	5 & 6	1364		14.95
TOTAL AVERAGE		1115	1117	19.62
Pit #2	1	835		22.48
	2	1260		13.20
	3	833		19.84
	4	1157		16.73
	5	935		32.01
TOTAL AVERAGE		1004	991	20.85
Pit #3 Destroyed by fire				
Pit #4	1	852		17.82
	2	1009		14.39
	3	1108		29.04
	4	1213		25.79
	5	1066		22.95
	6	862		17.12
TOTAL AVERAGE		1018	987	21.19

ⓧ Based on weight of refuse compacted in a field measured volume for each lift of placed refuse.

* Based on survey elevations taken on bottom of empty pit and top of final lift for volume determination. Total weight of all refuse in pit included in determination.

ASSESSMENT OF MUNICIPAL SOLID WASTE DISPOSAL IN SALINE/MARSHLAND ENVIRONMENTS

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ABSTRACT

The research design, data acquisition methodology and preliminary results of assessment of the environmental impacts of municipal solid waste disposal in coastal saline environments is described. Secondary data were gathered via computerized searches of several data bases covering both the published scientific literature and unpublished fugitive literature. None of the literature specifically addresses the problem at hand, but analysis of approximately 3,000 citations delivered revealed 217 useful documents.

A questionnaire mailed to 186 state solid waste and pollution control agencies produce meaningful responses from 18 states. At least one hundred forty seven sites were identified, of which at least 70 are still in operation. Some form of environmental monitoring has been performed at 70 sites.

INTRODUCTION

The forty-eight contiguous states are located on the Atlantic and Pacific Oceans, as well as on the Gulf of Mexico. When Alaska, Hawaii, and Puerto Rico are considered, this range expands to include the Arctic Ocean, Bering Sea, and the Caribbean Sea. The physical and biological conditions, therefore, range from arctic to tropical along the 86,500 miles (144,300 km) of U.S. shoreline which are subject to tidal fluctuations. This shoreline encompasses part of twenty-two states, as well as the Commonwealth of Puerto Rico (U.S. possessions such as the Virgin Islands, Guam, Marshall Islands, etc. were not considered within the scope of this study).

In the past, state and local governments, as well as many private citizens, viewed the coastal zone as a low productivity, low value environment. Since a very high percentage of our population is located adjacent to coastal areas, these low-lying lands have been used as depositories for solid waste. As undeveloped land became increasingly scarce, municipal solid waste was used as landfill to create additional dry land along the coasts. However, in recent years coastal lands have been formally identified as having significant ecological, flood control, water purification, and aesthetic values. Attempts have been made to calculate the economic value of particularly high productivity environments such as salt marshes.

This recognition has lead to several legislative and executive actions designed to protect these values. The state and federal environmental protection acts have sought, with varying degrees of success, to integrate environmental considerations into the governmental decision-making process. The Clean Water Act requires permits to dredge or fill in wetlands or waters of the U.S., and to discharge pollutants to surface waters. The Coastal Zone Management Act encourages integrated state planning and management in the coastal zone. State wetland protection acts strictly regulate modification and filling in some states.

Two major federal actions within the last three years have directly affected the use of coastal areas for municipal solid waste disposal. Regulations promulgated under the Resource Conservation and Recovery Act of 1976 (P.L. 94-580) have set up criteria for acceptable solid waste disposal sites. Wetlands are considered to be among a group of environmentally sensitive areas whose use is to be avoided. As the result of a determination that leachate generation constitutes discharge of a pollutant, National Pollutant Discharge Elimination System (NPDES) permits will be required, for which there will be a presumption against issuance. Those coastal areas which are not wetlands are regulated as floodplains. Disposal sites in such areas must be protected to the level of the 100 year storm. This provides an economic sanction against disposal in coastal saline environments. The other federal restriction operates in a more indirect manner. Executive Order 11990 (May 24, 1977) was issued "to avoid to the extent possible the long and short term adverse impacts associated with the destruction or modification of wetlands and to avoid direct or indirect support of new construction in wetlands wherever there is a practicable alternative." The order requires leadership action by federal agencies to avoid destruction or degradation of wetlands and their natural values, including avoiding direct or indirect support of construction on wetlands, unless there is no practicable alternative, and provided that all practicable mitigating measures are taken. Recent EPA implementing regulations require explicit justification of wetland projects, including demonstration that no practicable alternative exists.

As part of a larger study for the Solid and Hazardous Waste Research Division of EPA, Management of Resources and the Environment (MRE) is undertaking a secondary data search and assessment of the history and environmental impacts of municipal solid waste disposal in coastal saline environments. Disposal of sewage sludge, dredged material, cellar dirt, etc. are outside the scope of this study, except to the extent that they may be admixed with domestic and commercial refuse. (A less technical, although more appropriate term, would be common garbage.) Likewise, this study is not concerned with deep ocean disposal, but rather with material deposited on or adjacent to the immediate shoreline.

During the late 1960's and early 1970's, in an attempt to define the nature of the environmental problems facing us, many overview pollution studies were designed and executed. Wastler and Wastler (1972), of the then Federal Water Quality Administration, identified eight dominating environmental factors of estuarine zones in the United States. These included:

(1) The width and benthic character of the Continental Shelf

(2) The nature of associated currents

(3) The coastline structure

(4) The associated river flow

(5) The extent and quality of sedimentation

(6) The associated climate

(7) The associated precipitation

(8) The nature of the tide

By adding two additional general areas of concern, (9) leachate generation and movement, and (10) ecological impact, the study goals were drawn to specifically focus on solid waste disposal in saline environments and thus reinforce the need for a broad interpretation of potential and real environmental impacts.

STUDY DESIGN

Recognizing that published works normally provide a good base of data, we performed a preliminary and very limited literature search in order to test the specific procedures which could be used, and to add to our understanding of the study problem. As a result of that preliminary search, it was clear that, as initially suspected, relatively few specific articles are available in the scientific literature. We concluded that "fugitive literature," such as engineering reports and field inspection reports, would have to be diligently sought in order to effectively pursue case studies which should augment the more accessible literature, which is less specific.

For example, the Government of the Virgin Islands considered the creation of a marine landfill of approximately twenty acres in Stalley Bay on the south shore of St. Thomas. Requiring Army Corps permitting, with review and comment by EPA, the project involved diking and the disposal of municipal solid waste directly into the impounded marine environment. Predictions of a flow net analysis of leachate movement through the permeable dike and under the dike, and an evaluation of the probable ecological impact also were prepared. This type of information, which is current, site specific, and has meaningful data, would not normally surface in a standard literature search. Another complicating factor is the cross-disciplinary nature of the information desired. Use of conventional biological or oceanographic data bases would probably overlook valuable research.

Compilation of information from coastal communities relative to solid waste quantities and attitudinal responses requires an understanding of past solid waste management practices. Except for larger communities, the task of disposing of solid waste has been relegated to a low position in municipal operations. As such, record keeping and data gathering has historically been very limited. The extraction of this information from coastal states and towns would require a creative and vigorous investigation in order to assure a complete recovery of available information.

With these factors in mind, a strategy was developed to allow for a cost-effective extraction of relevant information on the history, environmental impacts, and governmental attitudes toward municipal solid waste (MSW) disposal in coastal saline environments. First, the published literature would be searched via computerized keyword access to such data bases as BIOSIS, Oceanic Abstracts, etc. Then the cross-disciplinary and gray or fugitive literature would be accessed via Environline, Endex, NTIS, WRSIC and SWIRS. Cross indexing via the Smithsonian Scientific Information Exchange and Scientific Citation Index and monitoring of Current Contents was proposed as a further check and to help combat the inevitable lag between publication and entry into computerized data bases.

In order to extract unpublished information and state/municipal attitudes, all governmental agencies with responsibility for regulating, planning, or monitoring coastal waste disposal would be queried directly through a mailed questionnaire.

COMPUTERIZED SEARCHES

Based on the size of the data bases, literature covered, degree of overlap (or non-overlap), timeliness, and cost, the following data bases were accessed:

Water Resources Scientific Information Center (WRSIC)

Solid Waste Information Retrieval System (SWIRS)

BIOSIS (Biological Abstracts)

Engineering Index (Endex)

Oceanic Abstracts

National Technical Information Service (NTIS)

Datrix

Pollution Abstracts

Enviroline

EIST

Smithsonian Scientific Information Exchange (SSIE)

Ph.D. Dissertations

Depending on the data base, the printouts included various amounts of information. At the least, the complete bibliographic citation, including title, author, journal volume and page (if applicable), data, and place of publication were given. Sponsoring agencies of contract research are usually specified. In addition, some entries included pagination, key words, descriptors, abstracts, or short annotations.

The SSIE data base is slightly different from all the others. The Smithsonian Scientific Information Exchange maintains a file of over 200,000 ongoing and recently completed basic and applied research projects sponsored by over 1300 Federal, state and local agencies, private foundations, and universities. The individual records in the file consist of: project title, supporting organization name and number, principal investigator, performing organization, project summary, reporting period and (usually) funding level. The purpose of using this file was to locate workers with active research interests in the field of waste disposal in aquatic environments. These individuals were then contacted by letter to see if their research, or any of which they were aware, had yielded results which would be of use to the study.

Several manual searches of non-computerized data bases were also conducted. In this way, the files of the then Office of Solid Waste Management and Planning, MERL, Yale University Library of Forestry and Environmental Sciences and MRE's technical library were also checked.

RESULTS

The literature searches confirmed that there is a large body of literature on solid waste disposal in the open ocean environment, impact of sewage sludge, sewage effluent and dredged material disposal in both near shore and open ocean environments, modification of coastal wetlands, terrestrial solid waste disposal, and bibliographies of same. What was lacking, however, was published information on the environmental impact of disposal of municipal solid waste in saline environments. A decision was made to recover all citations from which information could be extrapolated to the problem at hand.

As a result of the entire literature search effort, over 3,000 citations were generated. The breakdown of those deemed useful and which were acquired is presented in Table 1.

Table 1: Document Recovery via Literature Search

Data Base	Useful Citations	
WRSIC	50*	
SWIRS	23	
BIOSIS	9	
Endex	2	
Oceanic Abstracts	7	
NTIS	23	
Pollution Abstracts	19	
Datrix	1	
EIST	1	
Enviroline	9	
OSWMP	4	
SSIE	43	
Other manual searches	26	
	217	TOTAL

*does not reflect search update (1/79)

QUESTIONNAIRE SURVEY

As originally predicted, extraction of useful specific data on quantities, case histories, environmental impacts and government attitudes on municipal solid waste in saline environments required an aggressive data acquisition program aimed at state and local solid waste management and environmental protection agencies. The most effective methodology for contacting and eliciting this information was felt to be through direct mail contact. Accordingly, a questionnaire was designed which contained both closed and open-ended items. The closed items were designed to obtain factual response on specific issues, such as: have coastal saline environments been used as a repository for municipal solid waste in your state/city?; if so, has monitoring been conducted?; where are these sites located?; how long was the site used?; what type of waste was deposited?; what type of operation was involved?; etc. The open-ended questions were designed to determine agency attitudes toward the costs, benefits and public response to such waste disposal operations. The entire questionnaire is presented in the Figure 1.

Although provisions were included in the questionnaire for forwarding to a more appropriate agency, past experience had shown that more useful data would be collected in a shorter time period, if every attempt was made to determine the agency with primary responsibility within each state. Water pollution and solid waste management statutes were reviewed for each state in the study. Agencies with apparent responsibility were cross-checked with a file of U.S., state and local environmental officials to develop an accurate, up-to-date list of names and addresses.

RESPONSE

Initial response to the questionnaire was fair. One hundred and eighty six letters were mailed on August 1, 1978. 54 responses were received by 8/31/78, 9 more by 9/30/78, and a total of 67 by 10/31/78. Over thirty of these indicated that the questionnaire had been forwarded to a more appropriate agency. At this point, a second mailing was undertaken. In addition to contacting 123 non-respondents, a questionnaire was sent to thirty agencies not on the initial mailing list, who were referred to us by various respondees. By November

30, thirty-three more questionnaires were returned. The final results of the mail survey, in terms of response, are displayed in Tables 2 and 3.

DISPOSAL SITES

Responses were received from all states except Rhode Island. In addition, Virginia, North Carolina, Florida and Delaware provided no meaningful information. The information presented below is, therefore, based on data supplied by eighteen of the twenty-three states (including Puerto Rico) with borders on saline waters. Instances of municipal solid waste disposal were indentified in all of the eighteen states for which data were available; at least 147 separate sites were located. Environmental monitoring data of some sort are available at 77 sites in 11 states. These data vary from coliform counts to bioassays to water and sediment chemistry analysis. New Jersey, Maryland, California and New York have the most comprehensive data sets. At least one site was in operation for fifty years (ending in 1967). At least seventy sites located in coastal saline environments are still receiving municipal solid waste. These data are summarized in Table 4. Closure orders have been or soon will be issued for five of these sites. It should be noted that several states indicated that more complete data will be available as they complete their RCRA inventories.

STATE ATTITUDES AND LEGISLATION

Most states indicated that the RCRA solid waste criteria which severely restrict solid waste disposal in estuarine and coastal wetlands and floodplains will have no effect on their solid waste management programs. This was because preexisting legislation prohibited such siting. Alaska and Louisiana felt that the new regulation might have a serious impact due to a high percentage of regulated areas (the Alaskan response was based on the large areas of permafrost and/or muskeg environment and so was not directly responsive to the question.) Connecticut discourages new sites, but felt that vertical expansion of existing sites was preferable to creating new potential sources of pollution. They felt that discharge of leachate to a wetland should not be prohibited without site-specific review, due to the high population density and limited distribution of RCRA acceptable sites. In Massachusetts, new coastal MSW disposal operations are allowed only if no other option is feasible and if strict pollution control measures are incorporated into site design (their regulatory program closely parallels RCRA regulations). New Jersey allows permitted landfills in non-wetland coastal areas. No new sites have been approved in Oregon for several years, although there is no absolute prohibition. In Washington state, Indian nations currently use coastal disposal sites.

There are also several states which anticipate some impact as a result of restrictions on use of areas subject to inundation by a 100 year storm i.e., a storm of such intensity that its likelihood of occurrence is 1% for any particular year. Texas, Georgia, Connecticut and Alabama currently permit MSW disposal in floodplains. In the first three states, the present setback is to the level of the 50 year flood while in Alabama there are no floodplain restrictions. Connecticut will soon be switching to the 100 year storm standard.

CONCLUSIONS

Coastal saline environments have historically received large volumes of municipal solid waste. Although such use has tapered off dramatically in recent years, there are at least seventy such sites still in operation, and the actual number is probably at least twice that. Environmental monitoring data is available for many sites, although there has been virtually no standardization of procedures or parameters involved. Comparison among sites, therefore, will be difficult.

There have been major state and federal regulatory schemes adopted in recent years which restrict such waste disposal practices in the future. There has been, however, no systematic attempt to document the environmental impacts of such disposal. While sewage sludge dumping, sewage effluent disposal, dredge spoil disposal and other waste discharges to saline environments have been carefully categorized as to their impacts on water quality, sediment characteristics and benthic and free-living biota, municipal solid waste disposal has largely been ignored, except as it affects water quality.

The data that do exist are unpublished and to a large extent scattered throughout the the files of various state, regional and local solid waste and water pollution control agencies. They concentrate on short term leachate generation and migration parameters, with little or no attempt to determine if the leachate and/or landfill is affecting the surrounding ecosystem.

ONGOING RESEARCH

MRE will be conducting one of the most important aspects of the study this winter, although the results are not available at the time this paper is being prepared. As a result of the responses from the mail survey, several states were selected for more intense investigation. Criteria for selection included: broad biogeographical representation, a good-sized data set on several sites, and the inability of the responsible agencies to extract and supply the data in a timely manner. In this way, Connecticut, New York, New Jersey, Maryland, South Carolina, Louisiana, Washington and California were selected for visits. Together, these eight states contain 128+ sites, with monitoring available at 56 of them. This represents 87% and 73% respectively, of the known sites in the two categories. In addition, the literature search has been updated to include research conducted and/or entered into searched data bases in the 20-24 months that elapsed since the initial search in January, 1977.

Table 2. Response to Questionnaire Mailed August 4, 1978

STATE	Number of Agencies Contacted	Response Through: August 31,	September 30,	November 6,**	Total
Alabama	7	2(1)*	0	0	2(1)
Alaska	6	1	0	0	1
California	19	10(2)	1	1	12(2)
Connecticut	12	3(3)	0	0	3(3)
Delaware	6	1(1)	2	0	3(1)
Florida	11	3(3)	1 (1)	0	4(4)
Georgia	7	1(1)	0	0	1(1)
Hawaii	8	1(1)	1	3	5(1)
Louisiana	5	2***	0	1	3***
Maine	10	5(4)	0	0	5(4)
Maryland	8	6(2)	0	1	7(2)
Massachusetts	5	1	0	0	1
Mississippi	3	3(1)	0	1	4(1)
New Hampshire	6	1(1)	0	0	1(1)
New Jersey	7	1(1)	1(1)	0	2(2)
New York	8	5(4)	0	0	5(4)
North Carolina	8	0	0	0	0
Oregon	9	2(1)	0	0	2(1)
Puerto Rico	7	2***	1	0	3***
Rhode Island	6	0	0	0	0
South Carolina	7	3(1)	1(1)	1	5(2)
Texas	8	1(1)	0	1	2(1)
Virginia	6	0	0	0	0
Washington	9	0	1	1(1)	7

* numbers in parentheses indicate questionnaires which did not supply meaningful information, were forwarded to a more appropriate agency, or had already been answered by the appropriate agency

** date of mailing of follow-up questionnaire

*** includes one Return-to-Sender

Table 3. Response to Questionnaire Follow-up Mailed Nov. 6, 1978

STATE	Number of Letters Sent	Response Through:			
		November 30	December 31	January 31	Total
Alabama	6	3(2)*	1(1)	0	4(3)
Alaska	5	0	0	0	0
California	11	4(2)	1(1)	0	5(3)
Connecticut	12	2(2)	1(1)	0	3(3)
Delaware	4	3(3)	0	0	3(3)
Florida	10	3(3)	2(2)	0	5(5)
Georgia	7	2(2)	1	0	3(2)
Hawaii	6	1(1)	2(1)	0	3(2)
Louisiana	2	0	0	0	0
Maine	8	3(2)	0	0	3(2)
Maryland	4	1(1)	1(1)	0	2(2)
Massachusetts	4	1(1)	0	1	2(1)
Mississippi	1	2(2)	0	0	2(2)
New Hampshire	5	0	1(1)	0	1(1)
New Jersey	5	0	1(1)	0	1(1)
New York	6	0	4(2)	1	5(2)
North Carolina	8	2(2)	0	0	2(2)
Oregon	8	1(1)	0	0	1(1)
Puerto Rico	4	0	3(2)	0	3(2)
Rhode Island	6	1(1)	0	0	1(1)
South Carolina	5	3(2)	1	0	4(2)
Texas	4	0	0	0	0
Virginia	6	0	1(1)	0	1(1)
Washington	8	1(1)	1(1)	0	2(2)

* numbers in parentheses indicate questionnaires which did not supply meaningful information, were forwarded to a more appropriate agency, or had already been answered by a more appropriate agency.

Table 4: Municipal Solid Waste Disposal Sites in Coastal Saline Environments

STATE	KNOWN SITES	SITES WITH MONITORING	OPERATING SITES
Alabama	1	0	0
Alaska	many	0	many
California	40±	7±	20
Connecticut	5±	4	3
Delaware*	0	0	0
Florida*	-	0	-
Georgia*	-	0	-
Hawaii	1	1	1
Louisiana	4	4	4
Maine	5	?	3
Maryland	12	12	6
Massachusetts	8	8	7***
Mississippi	4	0	3
New Hampshire	0	0	0
New Jersey	40±	30±	22±
New York	11	7	7+
North Carolina*	0	0	0
Oregon	1	1	1
Puerto Rico	4	4	0
Rhode Island**	-	-	-
South Carolina	2+	1+	0
Texas	3	0	0
Virginia*	-	0	-
Washington	14+	1	3
	147+	77+	70+

* limited response

** no response

*** three sites under orders to close

Name of Respondent: ______________________________

Organization: ______________________________

Address: ____________________ Telephone: ______________

A. Have there been past, or are there ongoing instances of municipal solid waste disposal in coastal saline marsh areas of your state? _____ _____
yes no

B. Has monitoring or testing of environmental conditions of these operations ever been conducted? _____ _____
yes no

(If yes, please complete the following for each site for which monitoring or testing of environmental conditions has been conducted and for which data exists. Use additional sheets for multiple areas).

1. Identification of disposal site(s)(name/location) ______________________
2. Years of operation 19 through 19
3. Type of waste deposited ______________________
4. Type of operation and duration in years
   crush & cover____ burning_____ uncovered operation____
5. Describe environmental quality testing undertaken. Type of information gathered, when, agency performing testing. ______________________
6. Would you please forward copies of the program described in #5 above and the results obtained? ________
7. Responsible regulatory agency, group or indivdual having additional information: (include phone no. & address). ______________________

NOTE: The following questions may require the use of additional sheets for answering. Please staple them to this questionnaire.

C. What will be the effect of federal regulations severely restricting solid waste disposal in estuarine and coastal marshlands and flooplains on your solid waste management program? Please circle your response.

a. No. effect. This is already prohibited by state and/or local laws.
b. Some effect, especially in the following situations: ______________________
c. Significant effect, especially in the following situations: ______________________

D. If your agency has previously considered disposal in the coastal zone a viable alternative to deal with the solid waste problem, why has it done so? ______________________

E. What problems, both technical and with the public, have you encountered concerning solid waste disposal sites in estuarine and coastal marshlands? ______________________

F. What benefits do you see, both technical and from the public, from restricting solid waste disposal in estuarine and coastal marshlands? ______________________

Figure 1. Simplified Version of Questionnaire

SOIL COVER FOR CONTROLLING LEACHATE FROM SOLID WASTE

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ABSTRACT

Design of cover for controlling leachate from solid waste should utilize engineering procedures and concepts. Available soil classified in terms of USC and USDA systems can be ranked for assisting or impeding water flow, and on this basis soil choices can be made. Beyond this first step, however, there is a diversity of opportunity for exercising ingenuity in planning and in designing features to assure that a suitable cover is constructed.

INTRODUCTION

This paper summarizes some results of a study on cover for solid and hazardous waste.(1) The most common cover material has been soil, particularly indigenous soil at the disposal site, and such soil will continue to be preferred for many years even though synthetic and waste materials may be used in greater amounts in the future. Although some aspects of soil as cover have not been thoroughly investigated, much is standardized and well known from previous experience and practice in soil mechanics, agriculture, and soils construction.

SOIL CLASSIFICATION

Currently, soils used in solid waste disposal design and construction are classified according to either the Unified Soil Classification System (USCS) or the U. S. Department of Agriculture (USDA) System. The USCS is engineering oriented; soils are grouped according to gradation of particle sizes, to percentages of gravel, sand, and fines, and to plasticity characteristics. The USDA system classifies on the basis of texture only, i.e., percentages of gravel, sand, silt, and clay. Figure 1 presents relationships between the two systems based on laboratory tests on hundreds of soil samples.

MOVEMENT OF WATER THROUGH COVER SOIL

Infiltration

The Horton model(3) provides a simple conceptual representation of the general infiltration rate-time relationship (for a constant source of water at the ground surface). The maximum infiltration

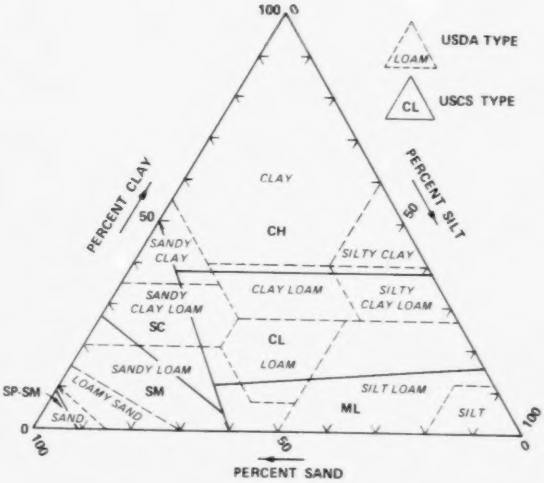

Figure 1. USCS superimposed on USDA soil chart(2).

capacity occurs at the beginning, and the rate decreases, first rapidly then more slowly, before reaching a stable minimum after as little as one hour. This curve can be conceptually applied to actual conditions. Thus, rainfall of intensity below the infiltration capacity curve (Figure 2) penetrates in its entirety.

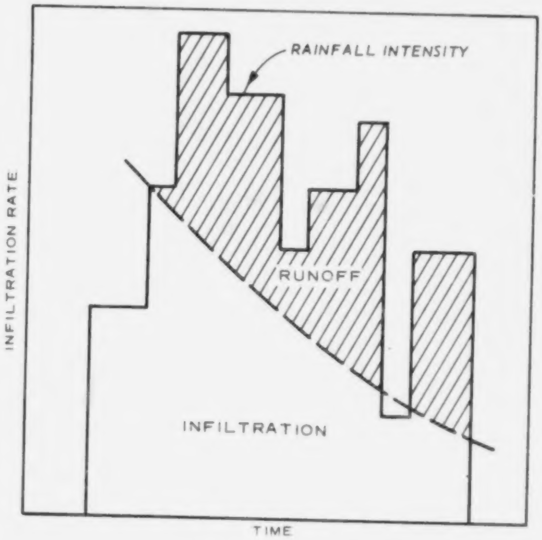

Figure 2. Infiltration capacity curve separating infiltration from runoff amounts in a rainfall event.

When the rainfall intensity exceeds the infiltration capacity, the excess over capacity accumulates on the surface and runoff occurs.

The most important factor affecting infiltration is usually the moisture content of the soil. The process is driven(4) by combined gravitational and capillary force acting downward. As the process continues, capillary pores become filled; and descending to greater depth, the gravitationally driven water encounters increased resistance due to reduction in large-pore space and sometimes also to a barrier such as clay. Hence, the initial, rapid reduction of infiltration rate (Figure 2) occurs. Low intensity periods may permit recovery of high infiltration capacity by providing time for elimination of surface water accumulations and time for internal drainage of gravitational water within the soil.

Percolation

The portion of infiltration which is important to leachate control is that continuing at a stable minimum infiltration rate (Figure 2) after soil pores are filled. This rate is approximated by the coefficient of permeability k of the soil. Various investigators(5) have shown that water flows through unsaturated soil according to a modified form of Darcy's law in which k is a nonlinear function of water content.

An important manifestation of the effect of k and its dependency on water content occurs in layered systems of contrasting soil types.(6,7) Where a wetting front, descending in a soil of relatively fine pore sizes, contacts a predominantly large-pored soil, the pore volume capable of holding water at the suction pressures existing at the wetting front is reduced, as is the water-filled cross section. Before the wetting front can advance, the suction pressure at that point must decrease until it is low enough to allow the pores to fill with water.

This hang up or delay continues until the upper layer approaches saturation, at which time the water is believed to descend along localized channels(6) rather than as a planar wetting front. Where the coarsely pored soil overlies the finer layer, a similar process reduces the rate of flow to an evaporation surface above.(7)

RATING AVAILABLE SOILS

From the discussions above, it is noted that the dominant characteristics of cover soils affecting water percolation and in turn leachate generation are permeability and water-holding capacity.

PERMEABILITY

Soils are rated in Table 1 for effectiveness in controlling percolation from I (best) to X (poorest) on the basis of k . Typical values of k are included for each soil type for clarification of

TABLE 1. SOIL RANKING FOR CONTROLLING PERCOLATION

USCS Soil	Description	Ranking For		
		Impeding	(k, cm/s)	Assisting
GW, GP	Gravel	X	(5×10^{-2})	I
GM	Silty Gravel	VII	(5×10^{-4})	IV
GC	Clayey Gravel	V	(10^{-4})	VI
SW, SP	Sand	IX	(10^{-2})	II
SM	Silty Sand	VIII	(10^{-3})	III
SC	Clayey Sand	VI	(2×10^{-4})	V
ML	Silt	IV	(10^{-5})	VII
CL	Silty Clay	II	(3×10^{-8})	IX
MH	Clayey Silt	III	(10^{-7})	VIII
CH	Clay	I	(10^{-9})	X

the absolute position of the particular rating. Selection or rating of soil at a site can be made quickly on this basis; e.g., an SM soil rates III and therefore is a reasonably good cover for assisting (rather than impeding) percolation, as might be desired in a system where leachate is recycled through the waste for faster stabilization.

WATER STORAGE CAPACITY

The importance of water storage in cover soil lies in the ability to absorb and at least temporarily hold infiltrated water that would otherwise percolate through. To quantify water-holding characteristics, one can formulate in terms of field capacity and saturated water content. Figure 3 presents approximate water storage capacities for the range of USDA soil types. Bound water, plant-available water, and gravitational water are distinguished for given soils according to availability for use by plants. Only gravitational water and plant-available water space contribute to water-holding capacity as far as the cover soil function of controlling percolation. For impeding percolation plant-available water space is best since it is not susceptible to drainage (percolation). Gravitational water space is only beneficial where a low-permeability layer slows free drainage at depth. Gravitational water space prevails in granular materials like sand. At the other extreme, gravitational water space is entirely subordinate to plant-available water space in clay.

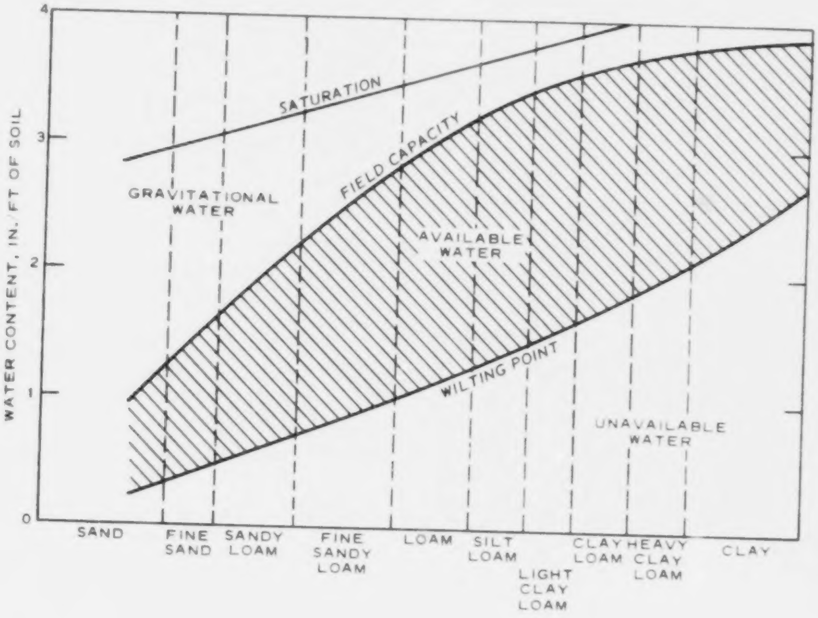

Figure 3. Water storage capacities for the range of USDA soil types.

ANALYSIS OF COVER FOR PERCOLATION

After the characteristics of the available soil have been evaluated as discussed above, an analysis can be undertaken to estimate the amount of percolation to be expected. This analysis provides a basis for making some refinements in design.

WATER BALANCE ANALYSIS

The water balance analysis has been proposed for solving the problem of estimating percolation.(8) Water balance equations assure continuity as follows:

PERCOLATION = PRECIPITATION - RUNOFF

- EVAPOTRANSPIRATION - CHANGE IN STORAGE

The procedure maintains a running account or tabulation(9) of water introduced to and removed from the system. Monthly increments are convenient and have been used most in the past.

Figure 4 shows graphically the water balance for 2 ft of loamy soil cover at Cincinnati, Ohio. The monthly rainfall means are based on a 25-year record. The basic curve, for infiltration, was obtained by subtracting a reasonable runoff percentage from each mean monthly precipitation amount. The actual evapotranspiration curve is obtained by adjusting the potential evapotranspiration estimates (derived according to Thornthwaite(9)) for the availability of water in storage; thus in dry months soil water in storage decreases and causes actual evapotranspiration to be less than potential levels.

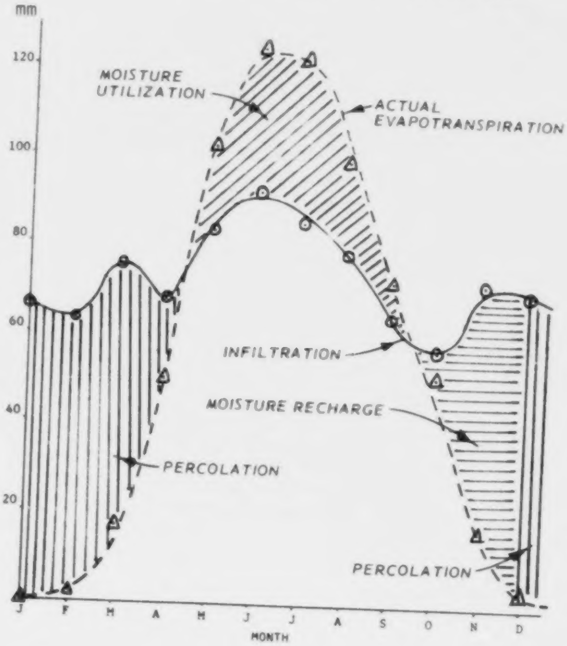

Figure 4. Monthly water balance in millimeters for 2-ft clay-loam cover at Cincinnati, Ohio(8).

Since water storage is a characteristic of the soil, then some flexibility for designing exists in the choice of soil and its thickness according to the water balance analysis. However, it does appear from this type of analysis(1) that thickening the soil cover to increase storage and thereby reduce percolation is not an efficient design procedure. The other important property k enters into the water balance equation above at only one place, in the runoff coefficient. Thus, soils with low k tend to have high runoff amounts subtracted at the start. However, permeability should also affect subsequent steps in water balancing and the fact that it does not provides a valid criticism of the technique discussed here.

CRITERION FOR ALLOWABLE PERCOLATION

Perhaps the greatest challenge in designing a cover for a humid area is in establishing what are tolerable limits of percolation. Attempts at solving this problem have been made previously.(8,10) Annual percolation has been estimated and used along with an assumed field capacity (1-4 in./ft) of the waste to estimate the time to field capacity of the waste cell. Upon reaching field capacity, the leachate will be generated at the annual percolation rate. This prediction is valuable in that it offers a starting point, albeit an oversimplified one. Experience shows that a leaking solid waste disposal site becomes a problem before field capacity. Water percolating through the cover usually becomes channelized, especially in heterogeneous solid waste such as from municipal sources.

A more realistic criterion for allowable percolation may be on the basis of what is the tolerable leachate rate. If a system is designed capable of collecting and retaining X in. of leachate, then that X value will dictate the allowable percolation rate through the cover. Accordingly, the designer may be advised to analyze, besides the average condition, a daily or even hourly water balance for the 10-year storm, etc. In this way, extreme throughflows will be incorporated in the design and will not surprise and overwhelm a system designed only for average percolation.

DESIGNING FOR INFILTRATION/PERCOLATION CONTROL

Several of the parameters influencing infiltration/percolation can be manipulated during design and construction. Presently, the usual intention is to impede the percolation of water, and the following guidance is oriented in that direction. For cases where the cover is intended to pass water freely, the appropriate procedure should also be apparent in contrast.

SELECTION OR BLENDING OF SOIL

The most direct means of designing a cover to impede percolation into solid waste cells is by use of fine-grained soil with inherently low k (Table 1). Conversely, where a need exists for relatively free percolation, the choice of cover soil should be among those with predominantly coarse grain sizes (and high k). Well-graded soils have lower k values than those with uniform gradation, i.e., principally of a small range of grain sizes (provided the median grain size is the same). Other factors, such as shrinking and cracking characteristics, and plant root development may complicate the choice.

If well-graded fine soil is not available nearby but coarse- and fine-grained soils are, consider blending for an acceptable product and increased source supply. Blending is effective only for increasing impedance to water movement since blending almost always results in a broadening of the grain-size distribution as compared with distribution of the component soils. Blending is usually an expensive operation, and a thorough review of other options should be made before proceeding on a large scale.

COMPACTION

As a general rule, the cover should be compacted. Compaction is effective in reducing infiltration and percolation since k values are sharply reduced. At municipal waste landfills, an effort of more than two passes of the spreading or compacting equipment may be impractical to monitor and regulate. Higher compactive effort seems appropriate for intermediate and final covers except as it will interfere with seeding grass. Care should be taken to assure that the water content during compaction does not greatly exceed the optimum percentage.

LAYERING

Layering (Figure 5) achieves effects not obtainable with a single material. A two-layer system should be sufficient for most cover and, in order to impede percolation, might consist of a layer of compacted clay of very low k value beneath a layer of silty sand to support vegetation, provide erosion protection, and help retain capillary water in the clay layer. The geometry and layer composition should be incorporated in the design on a site-specific basis after other options have been considered.

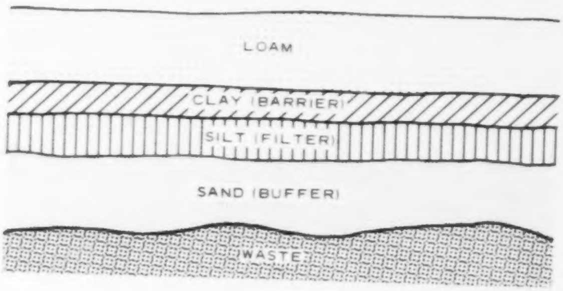

Figure 5. Design for a 2-ft layered cover over sand buffer on waste.

MEMBRANES AND BARRIERS

Numerous impermeable membranes and barriers are available to stop percolation where such a need has been established.

The costs are high, and cover systems incorporating such features are ordinarily only specified over hazardous wastes.

AVOID DISCONTINUITIES

An otherwise well-designed cover system may be compromised in time if deep cracks or offsets develop. Such discontinuities commonly form in association with differential settlement and, therefore, are aggravated by formation of depressions detaining some runoff. The first step to avoiding discontinuities comes in placement and compaction of the solid waste that forms the base. Heterogeneity should be reduced as much as possible; bulky objects such as hard frames or elastic tires should be kept from the upper part of the waste cell. A second beneficial procedure is to increase cover thickness sufficiently that anticipated differential settlement will not expose the waste cell to direct percolation. Ponding in associated depressions can be avoided by providing sufficient slope.

SURFACE SLOPE

Rainfall runoff increases with increasing slope of the surface; hence infiltration decreases. Since erosion also increases with increasing slope, the design of slope should balance the opposing considerations (where percolation is undesirable). On slopes of less than 3 percent, the irregularities of the surface and vegetation commonly trap and detain runoff. The value 5 percent has been suggested and used in grounds maintenance(11) as a first approximation of slope sufficient to promote runoff without risking excessive erosion.

WINTER GRASS

In mild climates, as in the southeastern United States, consideration should be given to growing both summer and winter grass to protect the cover from erosion and to extend the range of evapotranspiration to a year-round basis.

DRAINAGE

The first rule of surface drainage design for solid waste sites is to intercept and direct all water from outside the immediate area. Interception is accomplished by constructing a ditch or system of ditches on the uphill sides. Simply design the ditch to accommodate anticipated discharge, e.g., 10-year storm, according to an appropriate hydraulic formula.

On the cover itself discharge volumes are small, and the main concern is removing water quickly with minimal damage by erosion. A ditch slope of 2 percent is recommended.(11) Ditches should be grassed, and for proper mowing maintenance side slopes should not exceed 1 vertical on 3 horizontal or in cases of V-shaped channels, 1 on 6. Actually, flat-bottomed channels are preferred to V-shaped channels from the point of view of erosion.

Removal of cover soil water through buried drains offers promise for effective reduction of percolation into waste cells since such drains have been proven viable in engineering and agriculture. Drains require little maintenance and leave the surface in a natural and esthetically pleasing condition. A herringbone pattern of drains can be adapted easily to minor topographic irregularities as they are developed in landfilling.

FUTURE DIRECTION OF STUDY

A top priority need for refinement exists in the analysis of percolation amounts to be expected through soil cover. Once these amounts can be predicted accurately, there will be a basis and an impetus for adjusting soil composition or for refining designs accordingly. The study is currently proceeding toward that goal.

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LINER MATERIALS EXPOSED TO MSW LANDFILL LEACHATE

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ABSTRACT

This paper presents the results of immersing specimens of 28 different polymeric membrane liner materials in sanitary landfill leachate for 8 and 19 months. These 28 liners include variations in polymer, compound, thickness, fabric reinforcement, and manufacturer. The polymeric materials include butyl rubber, chlorinated polyethylene, chlorosulfonated polyethylene, elasticized polyolefin, ethylene propylene rubber, neoprene, polybutylene, polyester elastomer, low-density polyethylene, plasticized polyvinyl chloride, and polyvinyl chloride plus pitch.

The effects of the immersion were assessed by measuring the amount of swelling and the changes in such physical properties as tensile strength, elongation, modulus, hardness, and puncture resistance.

The results of the immersion tests generally confirm results previously obtained on membrane liner materials exposed for one year in simulated landfills. Specimens of chlorinated polyethylene, chlorosulfonated polyethylene, ethylene propylene rubber, and neoprene liners showed the greatest swell and loss of properties, although specimens of some of the ethylene propylene rubber and neoprene liners showed little swelling and loss of properties. Generally the change in properties of a given material appears to relate to the amount that it swelled.

Also reported are the results of the water vapor permeability testing of 27 membrane liners, the water absorption of a series of polymeric membranes at room temperature and at 70°C, and the retrieval and testing of samples of a six-year old membrane liner from a demonstration landfill.

A promising simple test for assessing permeability and physical properties of membrane liners for landfills is described. Test results are presented.

INTRODUCTION

This project was undertaken in 1973 with the primary purpose of assessing the effects upon a variety of liner materials of exposure to leachate from municipal solid wastes. At the time the project was initiated, the lining of landfills with various impermeable materials appeared to be a feasible means of preventing the migration of the leachate into the surface and groundwaters, resulting in their pollution. There was, however, little information available as to durability of the various available liner materials in such an environment.

There were considerable data as to the service lives of lining materials for containing water, but there was concern as to the long-term effectiveness of various liner materials in contact with leachate.

The primary effort of this project has been the exposure of selected lining materials in conditions which simulate actual use in sanitary landfills. Results of one year of exposure of these materials has been reported (1-5). The effects of one year of exposure were comparatively mild and inadequate with which to form a basis for making long-range projections of service lives. Consequently, the exposure periods were extended and the scope of the project was expanded to include studies on additional materials and an investigation of simpler methods of assessing the effectiveness and durability of the various liner materials.

This paper concentrates on the investigations of polymeric membrane liner materials and particularly on the effects of the total immersion of these materials for 8 and 19 months in leachate produced in the same simulated landfills used in this project. Other areas discussed in this paper are water absorption, permeability studies, and a description of a laboratory test for assessing the permeability and durability of membrane liners. The results of the recovery of a membrane liner from a demonstration landfill are also presented.

EXPERIMENTAL DETAILS

The conditions at the bottom of a landfill that must be simulated for the generation of leachate and for the exposure testing of liner materials are illustrated in Figure 1.

This schematic drawing of a liner includes the following features:

- Municipal solid waste.
- A porous soil cover on the liner on which waste can be dumped without damaging the liner and through which leachate generated in the fill can drain.
- The liner which may be from 30 mils to 3 ft in thickness, depending on the liner material used.
- Compacted soil on which the liner has been placed. The soil has been graded to give drainage to the leachate.
- Uncompacted native soil.
- Groundwater level, usually well below the landfill.

The design of the simulated landfill and leachate generator is illustrated in Figures 2 and 3.

Approximately 930 pounds of shredded municipal refuse was placed in each of the generators. The composition of the refuse is shown in Table 1; the amount of identifiable food waste was low.

TABLE 1. COMPOSITION OF REFUSE

Material	Percent
Water	12.2
Paper	53.6
Cloth	0.8
Plastics and rubber	4.9
Wood, garden, and food waste	5.1
Oils and fats	0.9
Metal	7.6
Glass, rock, and soil	14.9
Total	100.0

The leachate used throughout these studies was produced in the simulated sanitary landfills in which the primary liner specimens were also exposed. The average composition of the leachate at the end of the first year, when the first set of generators was disassembled and the liner specimens were retrieved and tested, is shown in Table 2.

TABLE 2. ANALYSIS OF LEACHATE[a]

Test	Value
Total solids, %	3.31
Volatile solids	1.95
Nonvolatile solids	1.36
Chemical oxygen demand (COD), g/l	45.9
pH	5.05
Total volatile acids (TVA), g/l	24.33
Organic acids, g/l	
Acetic	11.25
Propionic	2.87
Isobutyric	0.81
Butyric	6.93

[a] At the end of the first year of operation when the first set of liner specimens were recovered and tested.

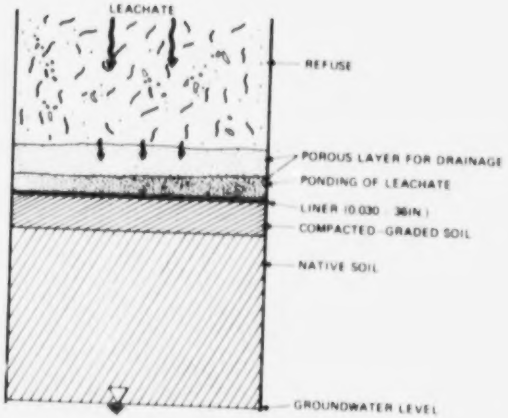
LEACHATE
REFUSE
POROUS LAYER FOR DRAINAGE
PONDING OF LEACHATE
LINER (0.030 - 36IN.)
COMPACTED-GRADED SOIL
NATIVE SOIL
GROUNDWATER LEVEL

Figure 1. Schematic drawing of the circumstances of the liner at the bottom of a solid waste landfill.

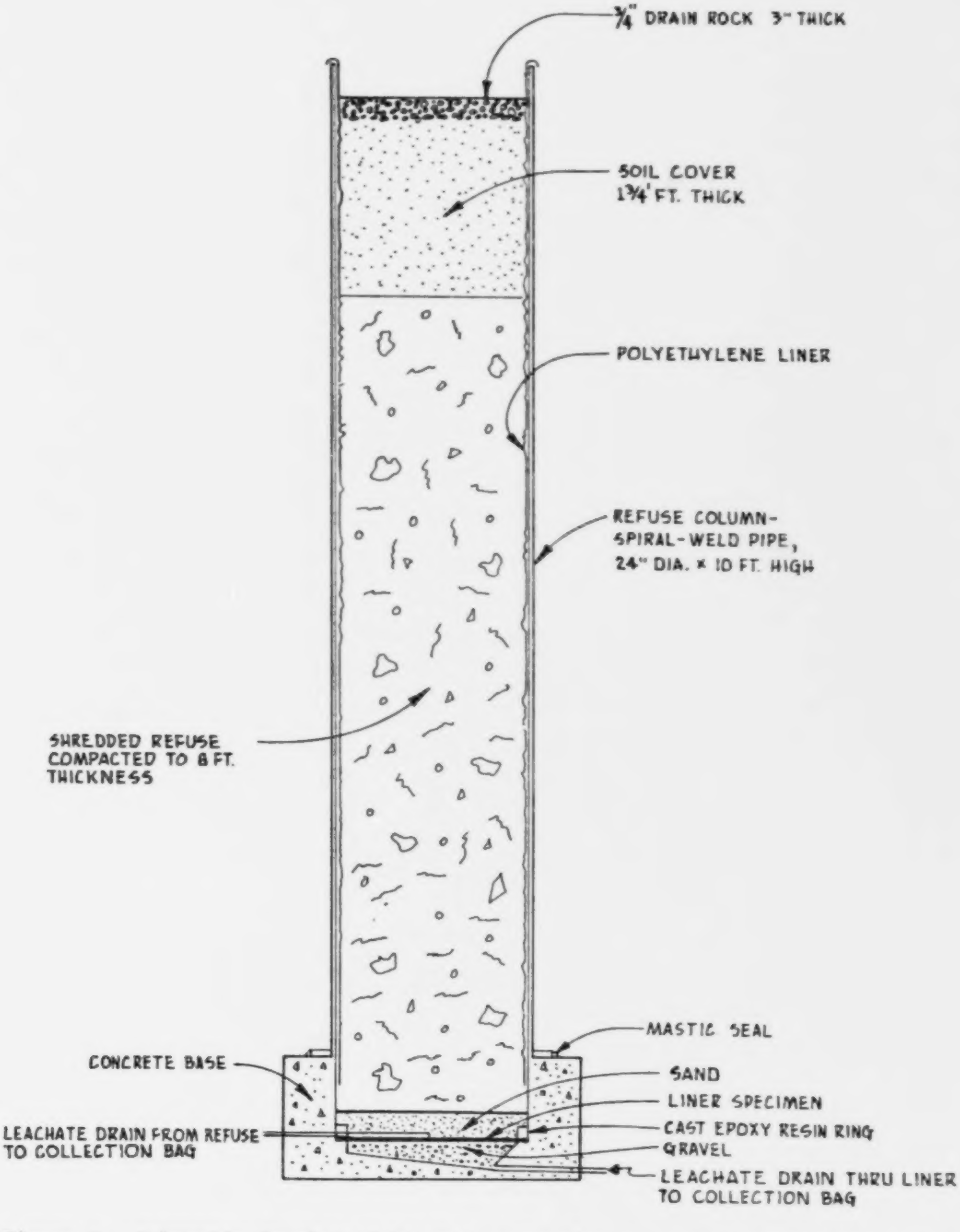
¾" DRAIN ROCK 3" THICK
SOIL COVER 1¾' FT. THICK
POLYETHYLENE LINER
REFUSE COLUMN-SPIRAL-WELD PIPE, 24" DIA. × 10 FT. HIGH
SHREDDED REFUSE COMPACTED TO 8 FT. THICKNESS
MASTIC SEAL
CONCRETE BASE
SAND
LINER SPECIMEN
CAST EPOXY RESIN RING
GRAVEL
LEACHATE DRAIN FROM REFUSE TO COLLECTION BAG
LEACHATE DRAIN THRU LINER TO COLLECTION BAG

Figure 2. Schematic drawing of leachate generator and cell in which the liner materials are being exposed to leachate under conditions simulating sanitary landfills.

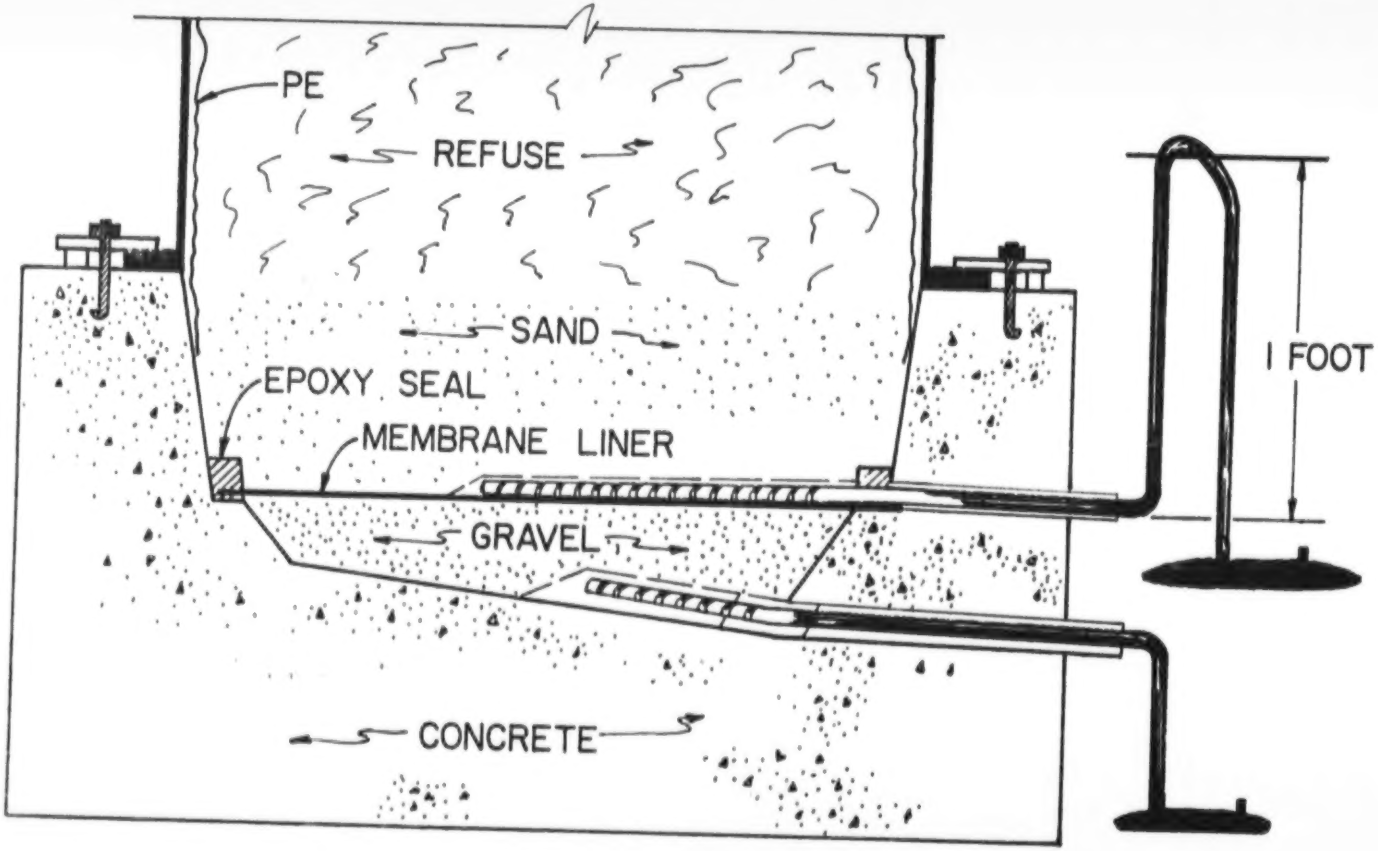
PE
REFUSE
SAND
EPOXY SEAL
MEMBRANE LINER
GRAVEL
CONCRETE
I FOOT

Figure 3. Detail of the base of the leachate generator showing the refuse, the sand cover, and the liner. One foot of leachate is allowed to pond on the liner. The additional leachate is continuously collected. Leachate that seeps through or by-passes the liner because of a failure of the seal is also continuously collected.

The strength of the leachate dropped as is shown in Figures 4 and 5. The compositions of the leachates from the 12 generators, which have remained in operation since the first group of liners were recovered and tested in November 1974, were initially quite uniform. However, in recent months the average strength of the leachates has dropped significantly and differences in compositions have developed. Nutrient (urea) is being added with the water to several generators to increase the strength, particularly of the volatile acids.

IMMERSION STUDY

The comparable results of the first year of exposure tests of the buried strips of the membrane materials with those of the primary liners indicated the complete immersion of a specimen in leachate would be a feasible method of exposing liners for test purposes (Table 3). For the generator exposures, the strips were placed in a curled position in the leachate-saturated sand directly above the primary liner specimens. In this position both sides of a strip specimen are exposed to leachate where only one side of a primary liner is exposed.

The availability of leachate from the simulated landfills made it possible to use the same fluid for immersing test specimens as is in the generators, thus allowing direct correlation between the immersion test and exposure at the bottom of the simulated landfills.

An immersion study was undertaken to evaluate membrane liner materials only, as admix materials do not lend themselves to tests of this type. The basic design of the immersion system consists of a series of six polyethylene tanks through which the leachate is allowed to flow at the rate of 14 ml per minute. The system is essentially airtight and the leachate is circulated once every 12 days. Each tank holds 14 specimens, 8 in. by 10 in. (Figure 6). Three sets of 28 specimens of each membrane were immersed in the leachate so that they could be removed in sets from the system after three exposure intervals. The 28 membranes of 11 different polymeric materials that were selected for testing consist of the following:

Type of material	Number of different liners immersed
Butyl rubber	1
Chlorinated polyethylene	3
Chlorosulfonated polyethylene	3
Elasticized polyolefin	1
Ethylene propylene rubber	5
Neoprene	4
Polybutylene	1
Polyester elastomer	1
Polyethylene	1
Polyvinyl chloride	7
Polyvinyl chloride + pitch	1
Total	28

The tests performed on the lining materials before exposure and at three subsequent intervals are:

- Weight.
- Dimensions.
- Tensile strength, in machine and transverse directions, ASTM D412.
- Hardness, ASTM D2240.
- Tear strength in machine and transverse directions, ASTM D624, Die C.
- Puncture resistance, FTM 101B, Method 2065.
- Volatiles at 105°C, ASTM D297.
- Specific gravity, ASTM D297.

The output of the 12 generators is blended and is added to the immersion system every four weeks, replacing part of the leachate in the system. During the first months of operation, the composition of the leachate before and after exposure was comparable to the average obtained by the mathematical averages of the compositions of the individual cells. Recently, the leachate appears to have changed considerably during the time it is in the immersion system, indicating a possible biological contamination of the immersion system.

After 8 and 19 months of immersion, the liner specimens were retrieved and tested. Selected results are presented in Tables 3 and 4 and Figures 7 and 8.

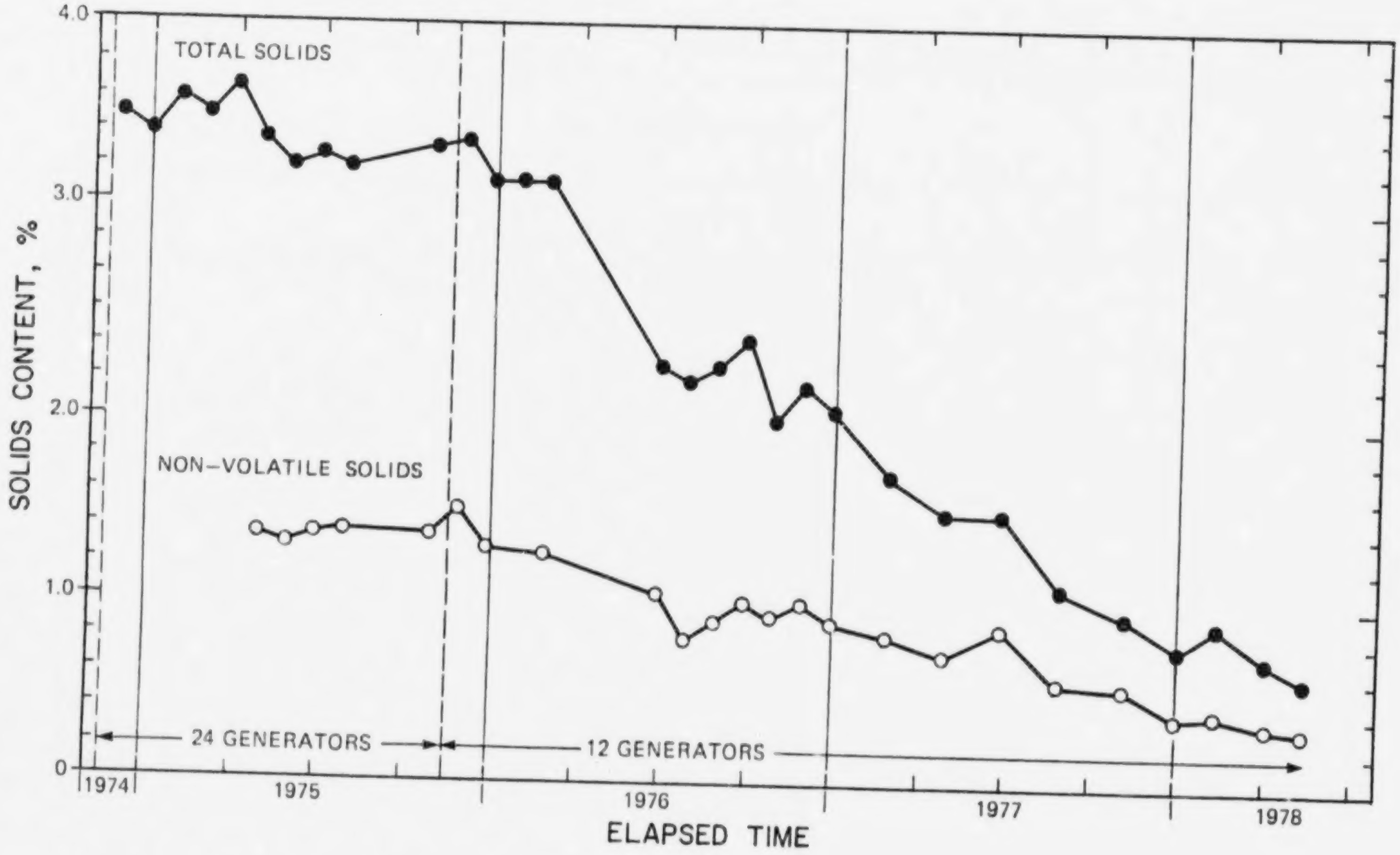

Figure 4. Average solids contents of the leachate produced in the generators, November 1974 - May 1978. The data for November 1974 - November 1975 are the averages for the leachate from 24 generators. Twelve generators were disassembled in November 1975 and, consequently, the data for December 1975 - May 1978 are the averages for the leachates from the 12 remaining generators.

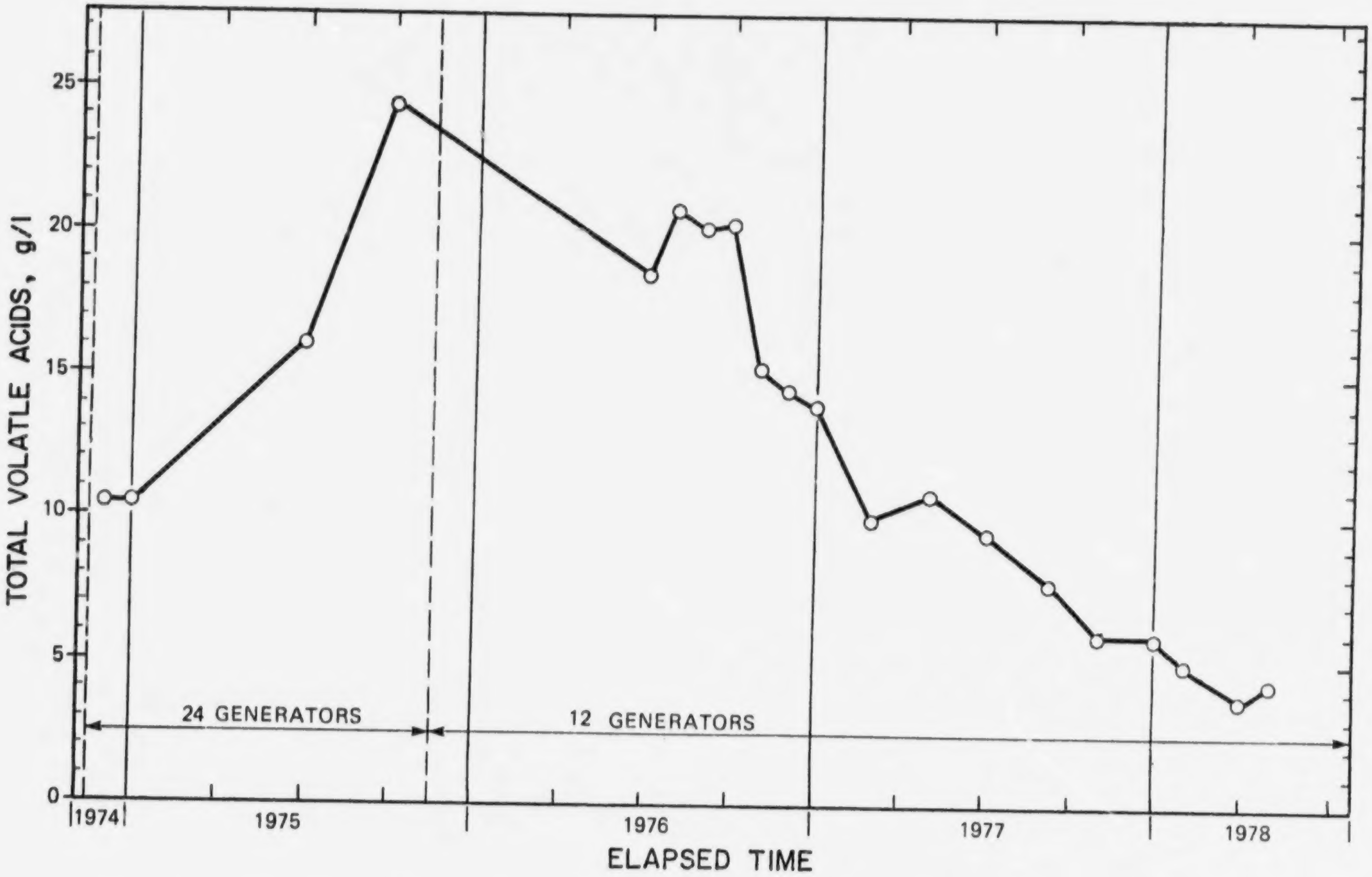

Figure 5. Average total volatile acids content (TVA), as acetic acid, of the leachate produced in the generators, November 1974 - May 1978. The data for November 1974 - November 1975 are the averages for the leachates from 24 generators. The data for December 1975 - May 1978 are the averages for the leachates from 12 generators.

TABLE 3. SWELLING OF POLYMERIC MEMBRANE LINERS BY LANDFILL LEACHATE - LEACHATE CONTENT OF EXPOSED MEMBRANE LINER MATERIALS IN PERCENT

Polymer	Liner no.	Exposed in simulated landfill for 12 months[a]		Immersion in leachate for 8 and 19 months			
				"Volatiles"[a]		From swelling test[b]	
		Primary specimens	Buried specimens	8 mo.	19 mo.	8 mo.	19 mo.
Butyl rubber	7	2.0	1.8	-	-	-	-
	44	-	-	1.4	2.6	1.8	3.4
Chlorinated polyethylene	12	6.8	9.0	7.9	14.4	8.7	15.0
Chlorosulfonated polyethylene	3	-	20.0	18.6	22.8	16.1	21.0
	6	12.8	13.6	12.1	14.9	12.0	13.9
Ethylene propylene rubber	8	-	5.5	2.9	3.8	5.7	7.0
	16	5.5	6.0	-	-	-	-
Neoprene	9	-	8.7	9.0	14.8	16.2	24.1
Polybutylene	20	-	0.3	-0.2	0.7	0.1	0.8
Polyethylene	21	0.02	0.3	0	0.2	0.6	0.7
Polyvinyl chloride	11	-	5.0	2.4	4.4	2.8	6.0
	17	3.6	3.3	2.3	4.4	3.2	3.2
	19	-	0.8	0.9	1.9	1.6	1.8

[a] Percent leachate in swollen liner as determined by "volatiles content" at room temperature and at 105°C.

[b] Calculated from the increase in weight of the immersed specimens.

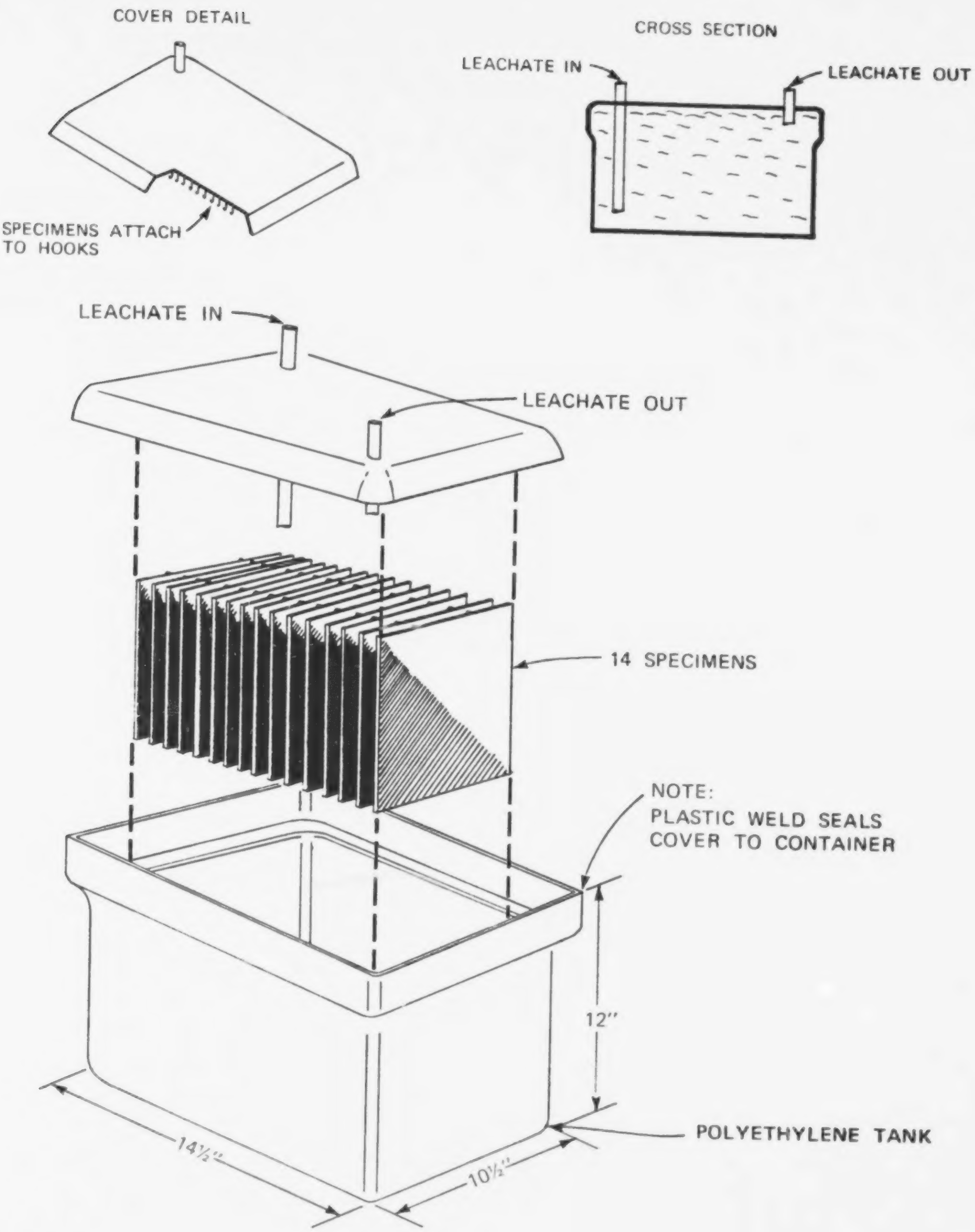
COVER DETAIL
SPECIMENS ATTACH TO HOOKS
CROSS SECTION
LEACHATE IN
LEACHATE OUT
LEACHATE IN
LEACHATE OUT
14 SPECIMENS
NOTE:
PLASTIC WELD SEALS COVER TO CONTAINER
12"
POLYETHYLENE TANK
14½"
10½"

Figure 6. Individual polyethylene immersion tank, showing method of holding specimens and the inlet and outlet for the leachate.

TABLE 4. RETENTION OF MODULUS[a] OF POLYMERIC MEMBRANE LINER MATERIALS ON IMMERSION IN LANDFILL LEACHATE

Polymer	Liner no.	S-200, psi			Retention % of original value	
		Unexposed	8 mo.	19 mo.	8 mo.	19 mo.
Butyl rubber	44	685	590	620	86	90
Chlorinated polyethylene	12	1330	1130	1190	85	89
	38	1205	1070	1090	89	90
	86	810	790	860	98	106
Chlorosulfonated polyethylene	3	735	395	335	54	46
	6[b]	1550	1775	2070	114	134
	85	1770	1360	1920	77	108
Elasticized polyolefin	36	970	1005	1050	104	109
Ethylene propylene rubber	8	690	870	860	126	124
	18	760	840	830	110	109
	83[b]	845	905	900	107	107
	91	855	775	790	91	92
	41	1040	1035	1030	100	99
Neoprene	9	1235	975	950	79	77
	37	1635	1630	1640	100	100
	42[b]	-	-	-	-	-
	90	1190	1245	1350	105	114
Polybutylene	98	3120	3160	3150	101	101
Polyester elastomer	75	2735	2790	2670	102	98
Polyethylene	21	1260	1340	1285	106	102
Polyvinyl chloride	11	2120	1845	1805	87	85
	17	1965	1570	1640	80	84
	19	1740	1545	1645	89	94
	40	1720	1560	1570	91	91
	67	1700	1570	1780	92	105
	88	2400	1900	2105	79	88
	89	2455	2370	2330	96	95
Polyvinyl chloride + pitch[c]	52	1020	870	880	85	86

[a] Average of stress at 200% elongation (S-200) measured in machine and transverse directions.

[b] Fabric reinforced.

[c] S-100 values given; original and subsequent exposed specimens failed at less than 200% elongation.

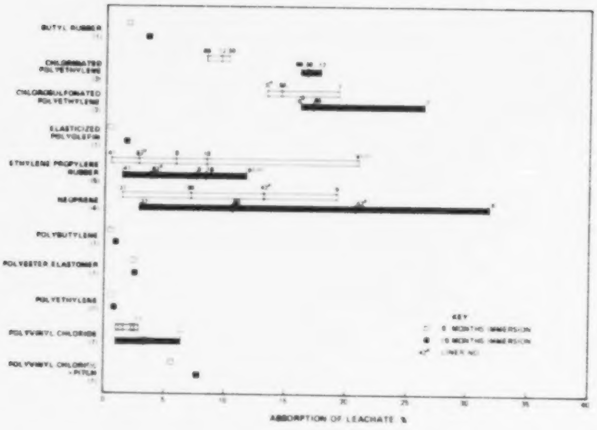

Figure 7. Swelling of membrane liners during immersion in leachate for 8 and 19 months.

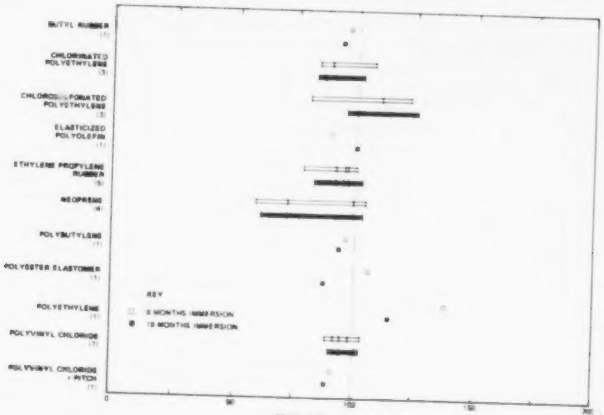

Figure 8. Retention of tensile strength of polymeric membrane liners on immersion in landfill leachate after 8 and 19 months. Tensile strength data were obtained by averaging the machine and transverse tests. The number of different liners of a given polymer that are included in the test is shown in parentheses.

The swelling that occurred during the immersion is compared in Table 3 with the swelling of samples of similar materials in the simulated landfills for 12 months. The buried and primary specimens generally swelled equally in the 12 months that they were exposed in the simulated landfills. In many cases, the 8 months of immersion were equal to the 12 months of exposure in the simulated landfills. On the basis of these data, it appears that the immersion in the leachate is quite comparable to the exposure in the simulated sanitary landfills.

In Figure 7 the data on swelling of the liners during the immersion in leachate is arranged by polymer type. The range of values for a given polymer is indicated by the bars for 8 and 19 months. The data show that, not only does the swelling vary from polymer-to-polymer, but also within a polymer type, because of compound differences. Chlorinated polyethylene, chlorosulfonated polyethylene, and neoprene showed the greatest variations within a polymer type. On the other hand, the seven polyvinyl chloride liner specimens varied only a few percent. In almost all cases the swelling values for the 19 months of immersion were higher than the eight-month values.

The effect on the tensile strength and modulus (stress at 200% elongation) of the same materials is shown in Figure 8 and in Table 4, respectively. The effects appear to be related to degree of swell which the specimens have undergone. Those specimens which swell little vary little in tensile strength and modulus. Again, the polyvinyl chloride membranes have a low spread in values and retained their original tensile and modulus. Overall, the polyolefins, such as polyethylene, polybutylene, and elasticized polyolefin, exhibit the low swelling and highest retention values.

The effect of immersion in leachate up to 19 months appears to have a relatively mild effect upon most of the membranes as shown for tensile strength and retention vs. immersion time in Figure 9.

The changes in properties, e.g. tensile strength, appear to be generally related to the amount of swell which the membranes have undergone during the immersion. This is illustrated in Figure 10, in which retentions of tensile strength have been plotted against the leachate absorption for the liner materials.

WATER ABSORPTION STUDY

The swelling of rubber or plastic compositions generally results in a reduction of mechanical properties and an increase in permeability. Severe swelling could ultimately cause the failure and nonperformance of polymeric membrane materials. During the first year of exposure in the simulated landfills, some of the specimens showed significant absorption of leachate and reduction in properties (2). Immersion of specimens in water resulted in comparable absorption, although the magnitude and the order of increasing swell were not the same as in leachate (Table 5).

The water absorption studies were extended using ASTM Test Method D570 (6). Absorption tests were run at room temperature and at 70°C. The objectives of these studies were to determine the correlation of swelling by leachate with that by water, which is the major constituent of the leachate. Also, water is a standard reference fluid for swelling tests. In addition, it was desired to determine whether the swelling of the liner materials would plateau, that is, come to equilibrium on extended immersion times. Results of water absorption of 11 membrane liners on immersion in water up to 100 weeks, both at room temperature and at 70°C, are presented in Table 6. The relative results for the two temperatures were generally the same; however, one of the major differences was with one of the polyvinyl chloride liners (#11), which yielded the lowest swell at room temperature, but was sixth at 70°C. The liner materials that had the lowest swell were the polyvinyl chloride, the elasticized polyolefin, and the ethylene propylene rubber; those that swelled the most were neoprene, chlorosulfonated polyethylene, and chlorinated polyethylene. The chlorinated polyethylene and the ethylene propylene rubber essentially reached equilibrium swell at 70°C in the 100 weeks. A polyvinyl chloride liner appeared to have reached equilibrium swell; however, it hardened because of loss of plasticizer. It appears that several of the other liner materials, particularly the neoprene and chlorosulfonated polyethylene, will continue to swell for considerable lengths of time.

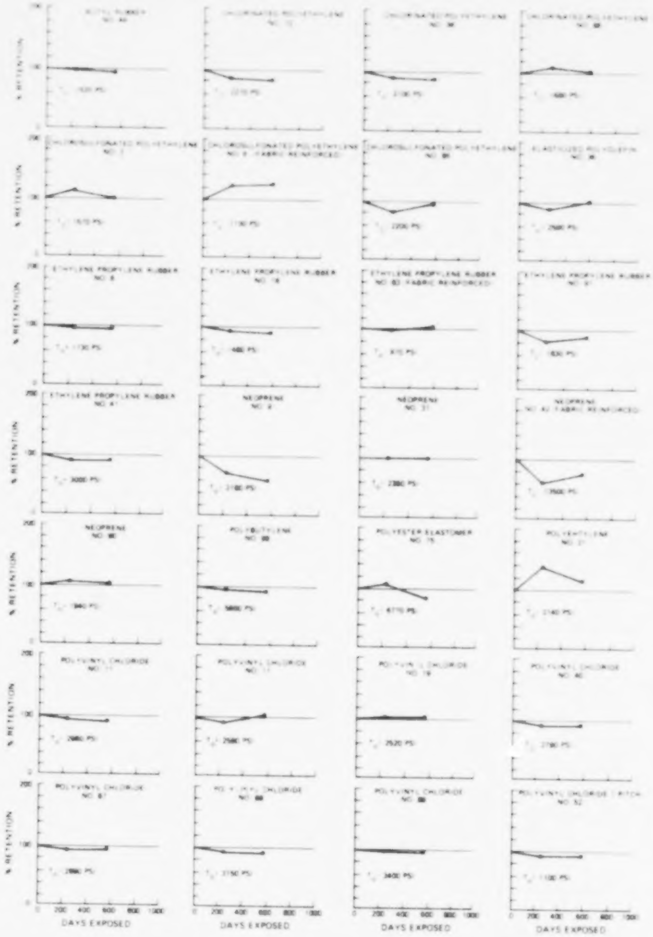

Figure 9. Retention of tensile strength of the individual polymeric membranes as a function of immersion time in landfill leachate. Tensile strength values based upon the average data obtained in the machine and transverse direction are used. Liner numbers as well as the initial tensile strength for each liner are shown. Data given for 8 months (240 days) and 19 months (570 days).

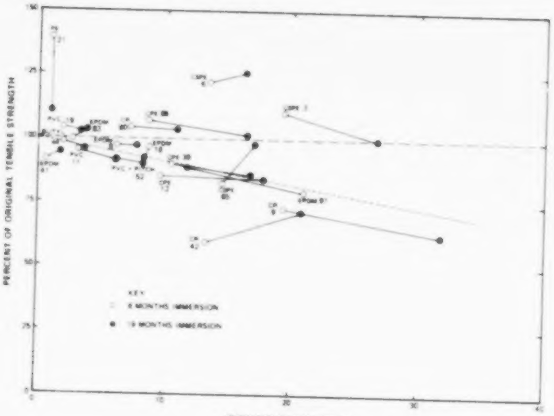

Figure 10. Retention of average tensile strength of membrane liner materials during immersion in landfill leachate as a function of swelling by the leachate.

TABLE 5. COMPARISON OF THE SWELLING OF MEMBRANE LINER MATERIALS IMMERSED IN WATER AND IN LEACHATE

Polymer	Liner no.	Swelling, % In water for 44 weeks	 In leachate for 32 weeks
Chlorosulfonated polyethylene	6	10.9	13.3
Elasticized polyolefin	36	0	0.1
Ethylene propylene rubber	8	1.6	6.0
Polyester elastomer	75	0.67	2.0
Polyvinyl chloride	11	0.70	2.9

TABLE 6. WATER ABSORPTION OF SELECTED MEMBRANE LINER MATERIALS AT ROOM TEMPERATURE AND AT 70 C[a]

		Water absorbed, % by weight								
		At room temperature				At 70 C				
Polymer	Liner No.	1 week	11 weeks	44 weeks	100 weeks	1 day	1 week	11 weeks	44 weeks	100 weeks
Butyl rubber	57	0.82	3.22	4.50	6.4	2.04	4.62	17.54	53.9	103.2
Chlorinated polyethylene	77	1.63	5.53	10.2	12.5	3.04	15.9	58.4	140.0	179.3
Chlorosulfonated polyethylene	6	3.44	6.97	10.9	16.3	5.68	22.1	131.0	245.6	370.5
Elasticized polyolefin	36	0.39	0.52	0.0	4.5	0.24	0.36	0.45	0.57	8.7
Ethylene propylene rubber	8	0.50	1.30	1.56	2.25	0.42	1.11	3.55	10.8	17.8
	26	1.20	1.84	1.49	2.56	0.74	1.44	4.52	11.20	17.4
Neoprene	43	3.80	13.62	37.8	75.1	3.89	14.1	107.0	240.0	(b)
	82	2.43	8.29	18.5	32.1	2.49	8.11	47.4	191.4	295.0
Polyester elastomer	75	1.07	1.05	0.67	1.31	1.18	1.28	1.10	0.72	0.22
Polyvinyl chloride	11	1.29	1.10	0.70	1.25	1.51	5.59	12.13	39.2	87.4
	59	1.59	2.34	2.43	2.98	2.09	4.87	8.25	24.0	25.5[c]

[a] ASTM D570-63 specimens 1 x 2 in. in deionized water.
[b] Specimens began to disintegrate between 44th and 69th weeks.
[c] Specimens have become hard, indicating loss of plasticizer.

These tests are being continued for the duration of the project. Also, physical testing of exposed samples will be made to determine the effect on physical properties from the swelling in water.

A comparison is made of the swelling in water and leachate in Table 5 for five liners. The swelling in leachate for 32 weeks was greater in all cases than swelling in water for 44 weeks, indicating the possible effect of the organic acid content of the leachate.

PERMEABILITY OF MEMBRANE LINERS TO WATER VAPOR

As the basic function of a liner is to control the flow of leachate and prevent its entrance into the surface and groundwater, permeability to water and dissolved ingredients is its most important property. The great range in the permeability of materials which are potentially useful as liners for landfills makes it difficult to use a single laboratory test to assess this property. The polymeric membrane lining materials are essentially nonporous and, consequently, the mechanism for the transport of fluids through the membranes is determined by the solubility and the rates of diffusion on a molecular scale. This is also true of asphaltic membrane liners. On the other hand, the transport of fluids through admix materials and soils is determined by a viscous flow mechanism. The driving force which moves fluids through membranes is largely determined by concentration differences, whereas the transport through the admixes is determined by hydrostatic pressure. The use of permeameters similar to those used for admixes requires extremely long tests.

The polymeric membrane liner industry has been using vapor transmission, following ASTM E-96, Method BW (7), as a measure of the permeability of the membrane liners. In this test, a small water cup with a membrane specimen cover is placed in an inverted position in an environment having controlled temperature, humidity, and airstream (Figure 11). The loss in weight of the cup is followed as a function of time and the permeability calculated from the slope. This test is intended for those applications in which one side is wetted under conditions where the hydraulic head is relatively unimportant and moisture transfer is governed by capillary and water diffusion forces. In this test the driving force is supplied by the difference in the vapor pressure from the two sides of the membrane.

Water vapor permeability of 27 polymeric membrane liners is reported in Table 7. The polyester liner, which was the thinnest membrane in the series, with a thickness of eight mils, had the highest rate of water vapor transmission. However, its water vapor permeability, which corrects for thickness and is a property of the liner polymer, is in the same range as that of polyvinyl chloride compositions, some of which are considerably more permeable. There is a threefold variation from the lowest permeability to the highest in the case of polyvinyl chloride liners. As a group, the polyvinyl chloride liners are the most permeable; the most impermeable materials in this test series are elasticized polyolefin and butyl rubber, both of which have less than one tenth of the water vapor permeability of polyvinyl chloride. There is a considerable spread in the values for butyl rubber, indicating compound differences. The other liner materials are intermediate in water vapor permeability, with a possible twofold variation among them.

Increased test times, during which the membranes are swelled somewhat by the water, yield slightly higher permeabilities, indicating that their permeabilities will be greater on long-term exposure. Additional tests of swollen liner specimens are projected.

BAG TEST FOR ASSESSING MEMBRANE LINER MATERIALS

The water vapor method described above for assessing the permeability of membrane liner materials yields results that are probably satisfactory for comparing the relative merits of the various liners to water. However, leachate contains dissolved constituents, many of which would not pass through the liner. Also, the test conditions in the vapor transmission method are not comparable to the conditions that a liner would encounter at the base of a landfill. The driving force, in the case of the vapor test, is a difference in humidity on both sides of the membrane. Such a condition would not exist at the base of a fill. The driving forces

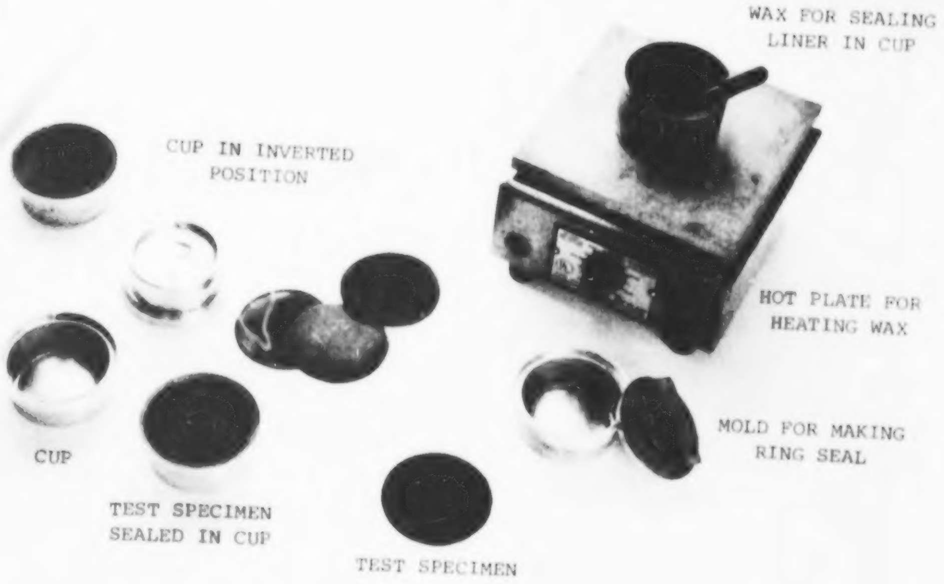
WAX FOR SEALING LINER IN CUP
CUP IN INVERTED POSITION
HOT PLATE FOR HEATING WAX
MOLD FOR MAKING RING SEAL
CUP
TEST SPECIMEN SEALED IN CUP
TEST SPECIMEN

Figure 11. E96 water vapor permeability cup and auxiliary equipment.

TABLE 7. WATER VAPOR PERMEABILITY OF POLYMERIC MEMBRANE LINERS, ASTM E-96, METHOD BW[a]

Polymer	Liner No.	Thickness mils	Thickness cm	Test time, days	Rate of water vapor transmission, $g/d \cdot m^2$	Water vapor permeance, 10^{-2} $g/d \cdot m^2 \cdot mmHg$ (metric perm)	Water vapor permeability, 10^{-3} $g/d \cdot m^2 \cdot mmHg \cdot cm$ (metric perm-cm)
Butyl rubber	22	73.3	0.185	49	0.097	0.75	1.39
	57	33.5	0.085	21	0.020	0.17	0.15
Chlorinated polyethylene	12	33.3	0.085	28	0.264	2.10	1.76
	38	32.3	0.082	21	0.361	2.90	2.35
	77	31.0	0.079	21	0.320	2.80	2.10
	86	21.0	0.053	28	0.643	5.10	2.72
Chlorosulfonated polyethylene	3	31.0	0.079	32	0.634	5.00	3.97
	6[b]	37.0	0.094	40	0.422	4.21	3.79
	55	35.0	0.089	42	0.438	3.47	3.09
Elasticized polyolefin	36	28.3	0.072	28	0.142	1.20	0.85
Ethylene-propylene rubber	8	67.0	0.170	28	0.172	1.40	2.52
	18	48.5	0.123	28	0.314	2.49	3.07
	26	38.0	0.097	28	0.327	2.80	2.72
	41	20.0	0.051	21	0.270	2.15	1.09
	83	37.0	0.094	28	0.190	1.50	1.42
Neoprene	9	61.0	0.159	21	0.237	1.90	2.89
	42	20.0	0.051	42	0.304	24.1	1.22
	43	31.5	0.080	28	0.448	3.90	3.12
	82	61.2	0.155	63	0.240	2.00	3.11
Polyester elastomer	75	8.0	0.020	21	10.50	91.0	18.2
Polyvinyl chloride	11	30.0	0.076	28	1.85	16.0	12.2
	17	20.0	0.051	35	2.97	24.0	12.0
	19	21.0	0.054	42	2.78	22.0	11.8
	40	32.5	0.083	42	4.17	33.0	27.3
	59	33.0	0.084	21	4.20	36.0	30.2
	88	20.5	0.052	35	2.94	23.0	12.1
	89	11.0	0.028	35	4.42	35.0	9.77

[a] Average temperature, 72°F; average relative humidity, 42%.

[b] Fabric-reinforced.

that exist are either hydrostatic pressure of a leachate on the liner or the difference in concentration of the leachate and the groundwater below. In the latter driving force, the osmotic pressure will force the fluid to go from a low concentration of dissolved constituents into a high concentration. A test has been devised in which the osmotic pressure will be the driving force for the movement of water and dissolved constituents across a membrane. This test involves the fabrication of small bags which can be filled with leachate or other test fluid and sealed and placed in deionized water (Figure 12). The flow of materials through the membrane can be measured by determining the weight increase of the bag and measuring the pH and electrical conductivity of the deionized water.

Initial experiments have been carried out with a 5% solution of sodium chloride and leachate. The composition of the leachate is given in Table 8. The test results are presented in Tables 9 and 10 for these fluids, respectively.

TABLE 8. CHARACTERISTICS OF LEACHATE IN BAGS[a]

Property	Value
Total solids, %	2.0[b]
Total volatile solids, %	1.1[b]
Chemical oxygen demand, g/l	35.7[b]
Total volatile acids, g/l	15.2[b]
pH	5.15[b]
Conductivity, μmho	11,500

[a] Amount of leachate in each bag is 100 ml. Samples were taken from the blend of leachates collected on November 8, 1976.

[b] Average value for the leachates taken from the 12 generators.

The ionic concentrations were calculated from the electrical conductivities using a plot of electrical conductivity and NaCl concentration for aqueous solutions (8). This calculation assumes the ions are principally sodium and chloride. The water permeabilities of the liner membranes are calculated from the respective increases in weight of the liner bags containing the test fluids.

Results with 5% NaCl solution in the bags of the various liner materials are presented in Table 8. The elasticized polyolefin and a polyvinyl chloride liner (#11) are the most impermeable to both water and the ions. The chlorosulfonated polyethylene was most permeable to the ions and the polyester elastomer was most permeable to water. The latter, however, was the thinnest (8 mils) compared with the others (>20 mils).

In the case of the bags containing leachate, it is again apparent that there was movement through the liner by both the water and the dissolved ingredients of the leachate (Table 10). An increase in electrical conductivity occurred, indicating the permeation of some ions from the leachate into the deionized water. Also, there was an increase in the weight of the bags containing the leachate, indicating permeation of water into the bags containing leachate. In this series, the elasticized polyolefin again yielded the lowest transmission of water and of dissolved components and the chlorinated polyethylene appears to be the most permeable. Also, one of the polyvinyl chloride liners (#17) appears to be quite permeable to ions.

RECOVERY AND TESTING OF SAMPLES OF A POLYVINYL CHLORIDE LINER FROM A DEMONSTRATION LANDFILL

Information from the field regarding the performance of artificial lining materials on long exposure to sanitary landfill leachate has been very limited. First, such use for these liners, particularly the polymeric membranes, is relatively new, dating from the early 1970's. Second, effective and economic methods of retrieving specimens and repairing linings at the bottoms of landfills have not been developed.

A demonstration landfill in Crawford County, Ohio, placed in the spring of 1971, was lined with a polyvinyl chloride liner. The liner from this landfill was relatively accessible as the total fill contained one lift of 8 ft of refuse and was about 12 ft deep, including the cover. This demonstration landfill had been designed to compare conventionally processed solid waste with rough and compacted wastes. The various types of refuse had been placed in essentially waterproof cells lined with plastic membranes. The effect of water content on consolidation and decomposition of the refuse was to be determined. However, all of the

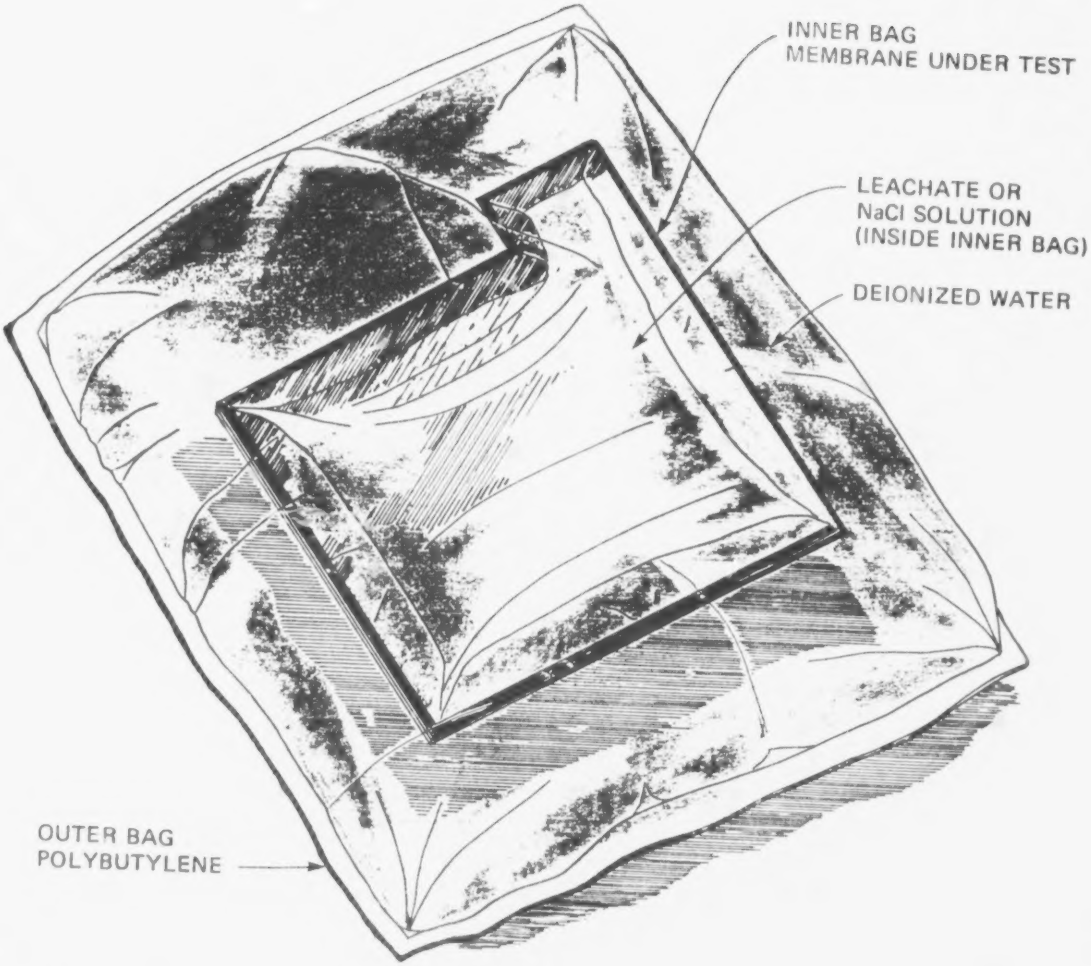
INNER BAG
MEMBRANE UNDER TEST
LEACHATE OR
NaCl SOLUTION
(INSIDE INNER BAG)
DEIONIZED WATER
OUTER BAG
POLYBUTYLENE

Figure 12. Schematic of osmosis bag assembly, showing inner bag made of membrane material under test. The inner bag is filled with leachate or 5% salt solution and sealed at the neck. The outer polybutylene bag, which can be easily opened, is filled with deionized water. The water in the outer bag is monitored for pH and conductivity; the inner bag is monitored for weight change.

TABLE 9. TESTS OF MEMBRANE LINER BAGS FILLED WITH 5% NaCl SOLUTION - PERMEABILITY OF LINERS TO WATER AND TO IONS DUE TO OSMOSIS

Polymer	Liner no.	Ion concentration deionized water[a], ppm	113 days immersion		200 days immersion	
			Ions[a], ppm	"Water" permeability[b] $g/m^2/s \times 10^{-7}$	Ions[a], ppm	"Water" permeability[b] $g/m^2/s \times 10^{-7}$
Chlorinated polyethylene	77	<1	13	21.6	17	18.8
Chlorosulfonated polyethylene	6	2	48	29.1	63	26.5
Elasticized polyolefin	36	<1	5	(c)	6	0.7
Polyester elastomer	75	<1	12	55.7	16	63.9
Polyvinyl chloride	11	<1	7	5.6	8	8.3
Polyvinyl chloride	59	<1	10	22.8	12	22.4
Blank	--	<1	4	--	4	--

[a] Concentration of ions (NaCl) in deionized water in outer bags; parts per million calculated from electrical conductivity measurements.

[b] Calculated from the weight increase in the inner bags containing the salt solution.

[c] Lost weight.

TABLE 10. TESTS OF MEMBRANE LINER BAGS FILLED WITH LEACHATE - PERMEABILITY OF MEMBRANES TO WATER AND TO IONS DUE TO OSMOSIS

Polymer	Liner no.	Original values		Values at 70 days			Values at 500 days		
		pH[a]	Ion conc.,[b] ppm	pH[a]	Ion conc.,[b] ppm	Water permeability,[c] g/m/s x 10^{-7}	pH[a]	Ion conc.,[b] ppm	Water permeability,[c] g/m/s x 10^{-7}
Chlorinated polyethylene	77	5.7	3	5.8	16	49.6	6.5	72	19.6
Elasticized polyolefin	36	5.1	2	5.0	5	(d)	4.5	10	0.9
Polyester elastomer	75	4.0	10	3.5	41	17.1	6.4	28	12.2
Polyvinyl chloride	11	5.8	3	4.4	18	12.1	6.0	18	4.6
Polyvinyl chloride	17	5.0	7	2.9	190	9.7	2.8	190	5.7
Polyvinyl chloride	59	5.7	3	3.8	35	28.6	6.3	9	5.0
Blank	--	5.5	<1	5.7	<1	--	4.3	6	--

[a] pH and conductivity of deionized water in the outer bag.

[b] Ion concentration of deionized water in outer bag calculated as NaCl from electrical conductivity measurements.

[c] Water permeability of liner calculated from weight gain of the inner bag containing leachate.

[d] Lost weight.

cells were flooded with water in a heavy rainfall just before the fill was closed. Thus, the original objectives could not be met and the project was terminated.

In view of the relative accessibility of the liner, the landfill was opened in May 1977, after six years, and the membrane liner was recovered. The cells appeared to have retained the water. The condition of the refuse did not appear to be typical of sanitary landfills. The odor was mild, and the refuse showed little deterioration. The samples of liners recovered from the cells appeared to be in excellent condition with little difference apparent between samples taken from the top of the cell, above the refuse, and those taken from the bottom, below the refuse. The top liner had been under 3 ft of clay cover, and the bottom liner had been under about 2 ft of clay and was on top of pea gravel. Both liners had taken the shape of soil and gravel without breaking. The depressions were as much as 6 in. deep in a 1 ft area in the exposed top liner.

Results of tests of samples taken from the top of the fill and from the bottom of the fill are given in Table 11. However, there was no sample of the original material available for comparison. The amount of swelling and the possible decrease in properties were within the range of values observed in the immersion testing for the seven polyvinyl chloride materials. The specimen taken from the bottom was probably not in direct contact with the leachate in the fill during the six years it was in place. The two-foot soil layer above the liner was a highly impermeable clay having a permeability coefficient 1.4×10^{-7} cm/sec. The clay layer and the weak leachate created a situation that is not typical of what is expected in a fullscale landfill.

ACKNOWLEDGMENTS

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TABLE 11. PROPERTIES OF POLYVINYL CHLORIDE LINER RECOVERED FROM A DEMONSTRATION LANDFILL IN CRAWFORD COUNTY, OHIO

Property	Test method	Liner from top of fill (#96)		Liner from bottom of fill (#97A)	
		Machine	Transverse	Machine	Transverse
Thickness, mils	--	30	--	28	--
Tensile strength, psi	ASTM D412	2665	2600	2550	2475
Elongation at break, %	ASTM D412	340	360	325	350
Set at break, %	ASTM D412	66	79	55	70
S-100, psi	ASTM D412	1290	1245	1185	1085
S-200, psi	ASTM D412	1830	1750	1785	1605
S-300, psi	ASTM D412	2245	2300	2400	2205
Tear strength, ppi	ASTM D624	374	370	343	341
Hardness (Duro A), instant	ASTM D2240	77	--	78	--
Puncture resistance, lb	Fed Std 101 -B-2065	41.4	--	37.3	--
Elongation, in.		0.66	--	0.65	--
Seam strength (shear), ppi	ASTM D413	49.5	--	45.5	--
Locus of failure		SE[a]	--	BRK[b]	--

[a] Break at seam.
[b] Break in tab.

FORECASTING PRODUCTION OF LANDFILL LEACHATE

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ABSTRACT

The results of several research projects are reviewed with respect to predictable trends in leachate composition. Two projects resulted in forecasting models which are briefly described. Potential applications of a technique to forecast leachate production are provided along with factors which are considered important in making the technique useful for design, operation and regulation of municipal refuse landfills.

Introduction

Disposal of municipal refuse to the land results in gaseous and liquid emissions which may have a detrimental effect on the environment and the public health. The severity of the effect will depend on the effectiveness of hydrogeological conditions at the disposal facility and on the effectiveness of control systems specially constructed to augment the site's natural control processes. The characteristics of the leachate (quantity, composition and production rate) produced by rainwater infiltrating the decomposing wastes greatly influence the effectiveness of natural control processes. The ability to estimate the leachate characteristics which the surrounding environment must accept has not received much attention, despite the fact that such information can be very useful in reducing the subjectiveness of environmental assessments and improving the design of disposal facilities. This paper is intended to describe the need for a leachate forecasting procedure, why achievement of such a goal seems plausible, and what factors are considered important.

Background

The practice of municipal refuse disposal by sanitary landfilling has been studied sporadically for several decades. Eliasson (1) characterized leachate from a few metropolitan New York landfills and estimated the moisture requirements for vigorous biological activity. Merz (2) using plywood bins placed in an operating landfill provided several months of data on leachate composition. The work by Merz and Stone (3) identified the importance of moisture content and temperature on the rate of waste decomposition which was estimated by measuring gas production.

The work of Fungaroli (4) yielded long-term information on leachate characteristics under simulated East Coast precipitation/infiltration conditions, and successfully described the movement of leachate in groundwater underlying a field-scale test cell containing several hundred tonnes of refuse. Hydraulic and long-term leaching characteristics of shredded refuse were also reported. The work of Burchinal and his co-workers emphasized the importance of microbial activity within a landfill (5-6). Temperature and moisture were identified as being very important in controlling the microbial activity; the importance of depth of leached waste was studied.

These and other studies indicated a potential for water pollution was created by leachate which generally originated from water percolating landfilled, microbially active municipal solid waste. A common factor of these early studies was intentional application of water to the "landfill." A common criticism at that time (late 60's) was that leachate was a laboratory anomally; that operating landfills which followed good practice (compact the refuse, cover daily, select and slope cover material to shed water) were not known to have created or threaten surface or groundwater contamination nor even if leachate was produced.

The Solid and Hazardous Waste Research Division (SHWRD), EPA installed a 395 tonne experimental landfill (Test Cell 1) at its Boone County Field Site (BCFS) to determine if application of good practice would produce a leachate and, if so, the composition. The volume of leachate collected was in general agreement with estimates made according to the procedures of Thornthwaite which were applied to landfills by Remson *et al* (7) and Fenn *et al* (8). Four additional experiments were installed at the BCFS: three 2 tonne, 0.91m diameter test cells (2A, 2B, and 2C) and one 72 tonne, 8.5m square test cell. The purpose of these test cells was to determine the replication of small-scale experiments and the comparable performance of large and small-scale test cells. Wigh (9) has presented a review of results from these five experimental landfills located at the BCFS.

Sensitivity of leachate composition to different infiltration rates was one of the objectives of a series of 19 experimental landfills constructed by USEPA at the Center Hill Facility in Cincinnati. There were constructed identical to the small scale test cells at the BCFS; results have been periodically reported by Jackson (10), Streng (11-12) Swartzbaugh (13) and Walsh (14).

Observations

A review of the BCFS results and other studies where the landfilled refuse has been subjected to single pass leaching of a batch of refuse. (No recirculation of leachate and no addition of fresh refuse) indicates leachate composition follows similar trends (Figure 1). Assay results, expressed as concentrations, tend to peak at the same time that the refuse begins to leach in equilibrium with infiltration, followed by a gradual decline to a low-level concentration which persists for a long time. The apparent relationship of the peak concentration and attainment of field capacity was noted by Wigh for a number of assays performed on the BCFS leachates (Table 1). Results of Pohland (15) suggest that experimental landfills which are rapidly brought to field capacity achieve peak concentrations only after the microbial activity has had sufficient time to influence leachate composition.

Constituents in leachate from any particular study are subject to variations (Figures 2&3) from sample to sample, but as a whole follow the general declining trend depicted in Figure 1. More consistent trends were noted when assays were represented as a function of cumulative leachate volume (Figure 4). The results of several different experimental landfills at the Center Hill Facility (Figure 5) followed a very similar trend, regardless of the rate of infiltration; the addition of per capita equivalent sewage sludge and 3% by weight of $CaCO_3$ had no significant effect on the general trend. The empirical results showed greatest variation during the initial leaching which represented leachate collected prior to and immediately after field capacity. The net throughput of water, measured as infiltration or leachate, is seen to be an important variable in describing leachate composition.

The mass of material released from these experimental landfills has also followed very consistent trends. Release of total solids (Figure 6) for an experiment receiving 813 mm/yr of infiltration decreased gradually as the volume of infiltration (leachate) increased. Comparable experiments, but receiving 406 mm/yr and in some cases with additive materials (initial water, sewage sludge, and $CaCO_3$), released total solids in similar patterns.

The consistency of such observations led some investigators to consider empirical modeling. Wigh (16) showed good agreement between BCFS - TC2 data and an equation which contained empirically determined constants (Figure 7). The peak concentration and the volume of leachate when the peak occurred were used to solve the equation; constants K_1 and K_2 were determined by trial and error. The resulting equation when applied to an identical experiment yielded a 3.5% over-estimate (@ 1200 mm) of actual, empirical results, and when applied to a comparable field scale experiment yielded a 9.2% underestimate (@ 1400 mm). The equation in its present form, however, is applicable only to systems built and operated in a comparable manner.

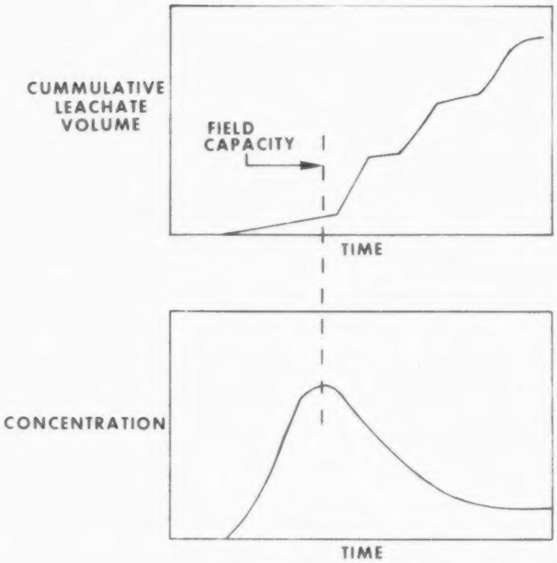

FIGURE 1. General response of leachate composition from a single pass leaching of a fixed amount of municipal refuse.

TABLE 1. PEAK CONCENTRATIONS AND FIELD CAPACITY[a]

PARAMETER	EXPERIMENT PEAK CONCENTRATION[b]	2A TIME[c]	EXPERIMENT PEAK CONCENTRATION[b]	2B TIME[c]
COD	57330	51	61600[d]	37
Total Kjeldahl-N	1560	51	1897	63
Ammonia-N	1035	65	1185	63
Ortho-phosphate	390	43	185	45
Sulfate	1306	51	2000	63
Sodium	1900	53	1700	49
Potassium	2225	49	2939	65
Chloride	2335	51	2343	59
Iron	1547	53	2902	37
Magnesium	486	51	617	63
Manganese	109	43	115	39
Calcium	2280	49	4000	39
Zinc	150	49	360	49
Hardness	7067	45	10575	37
Total Solids	46484	51	45628	49
Alkalinity	11535	51	13880	29
Acidity	6720	45	6843	49
Conductivity	17000	53	18000	53
Mean		50.2		49.3
Standard Deviation		4.9		11.4
Leachate		43		27
Field Capacity[e]		43		43

b. mg/l

c. Weeks since construction

d. Early peak, conentration later dropped and peaked again on 11/7/73 sample.

a. From Boone County Field Site Test Cells 2A & 2B.

e. From water storage calculations.

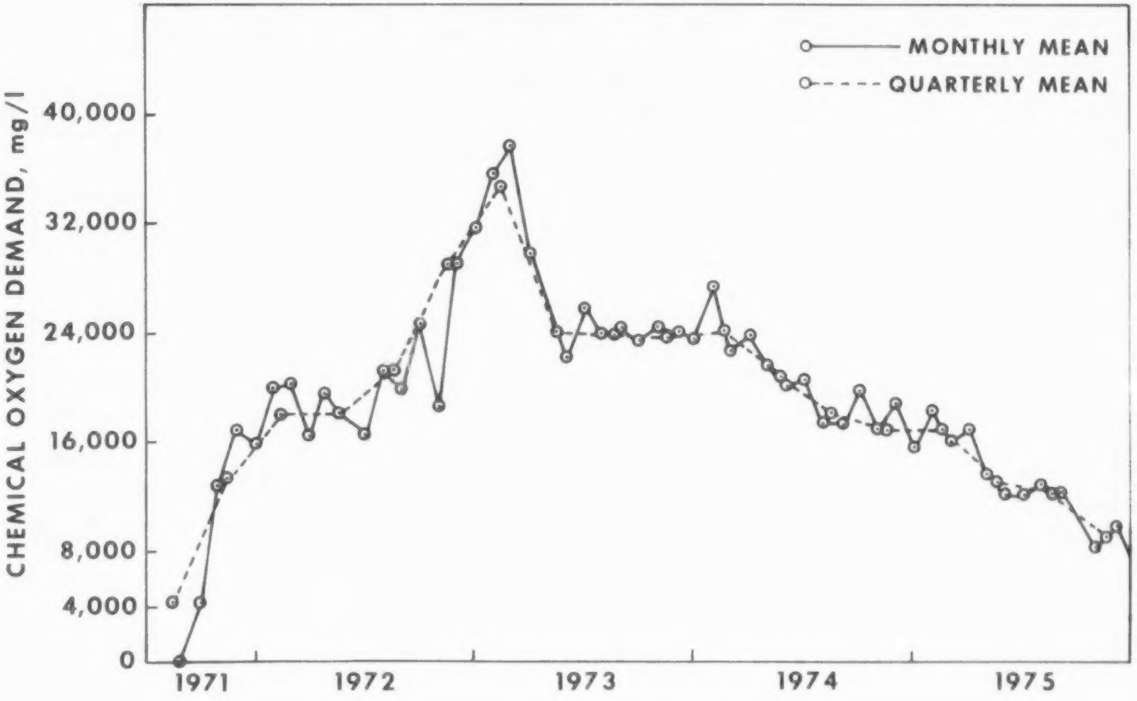

FIGURE 2. Boone County Field Site Test Cell 1 weighted mean chemical oxygen demand concentration.

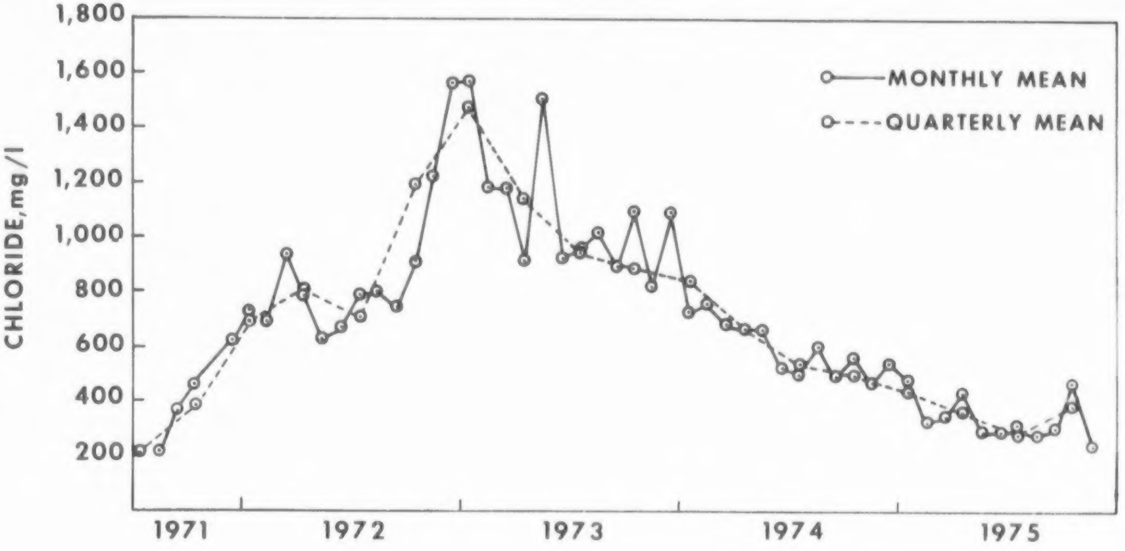

FIGURE 3.Boone County Field Sight Test Cell 1 weighted mean chloride concentration.

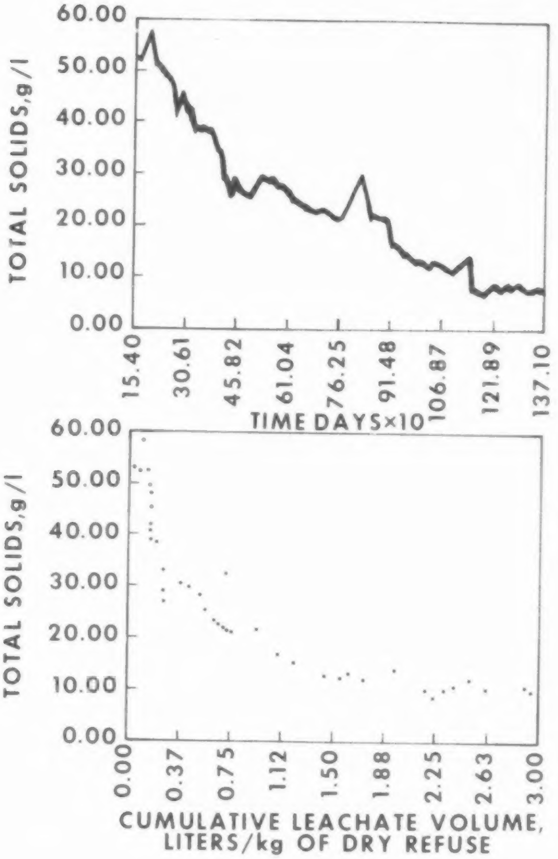

FIGURE 4. Total solids concentration from experimental refuse landfill leached @ 813 mm/yr.

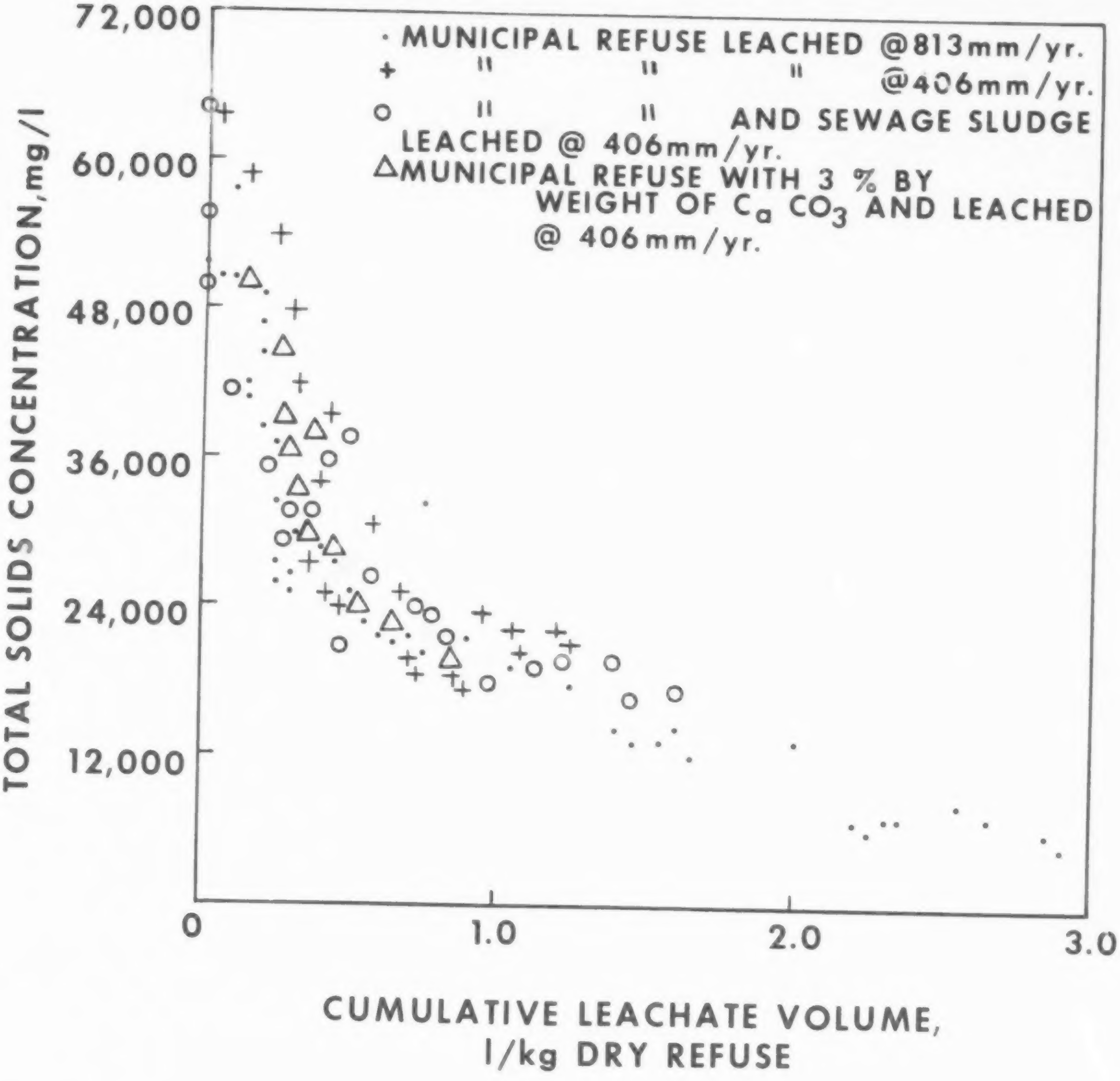

FIGURE 5. Total solids concentrations released from different experimental landfills.

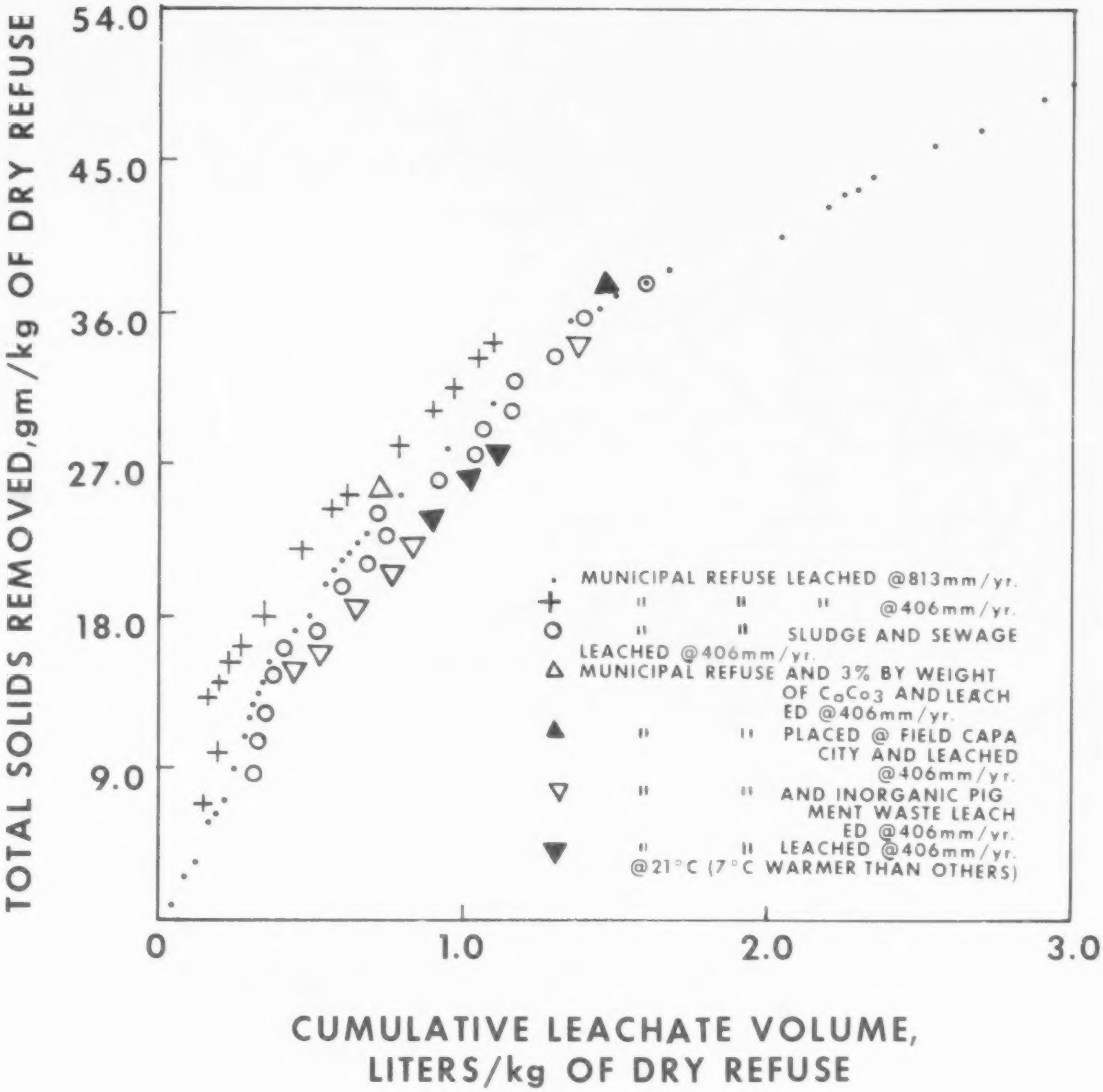

FIGURE 6. Total solids removed from different experimental landfills.

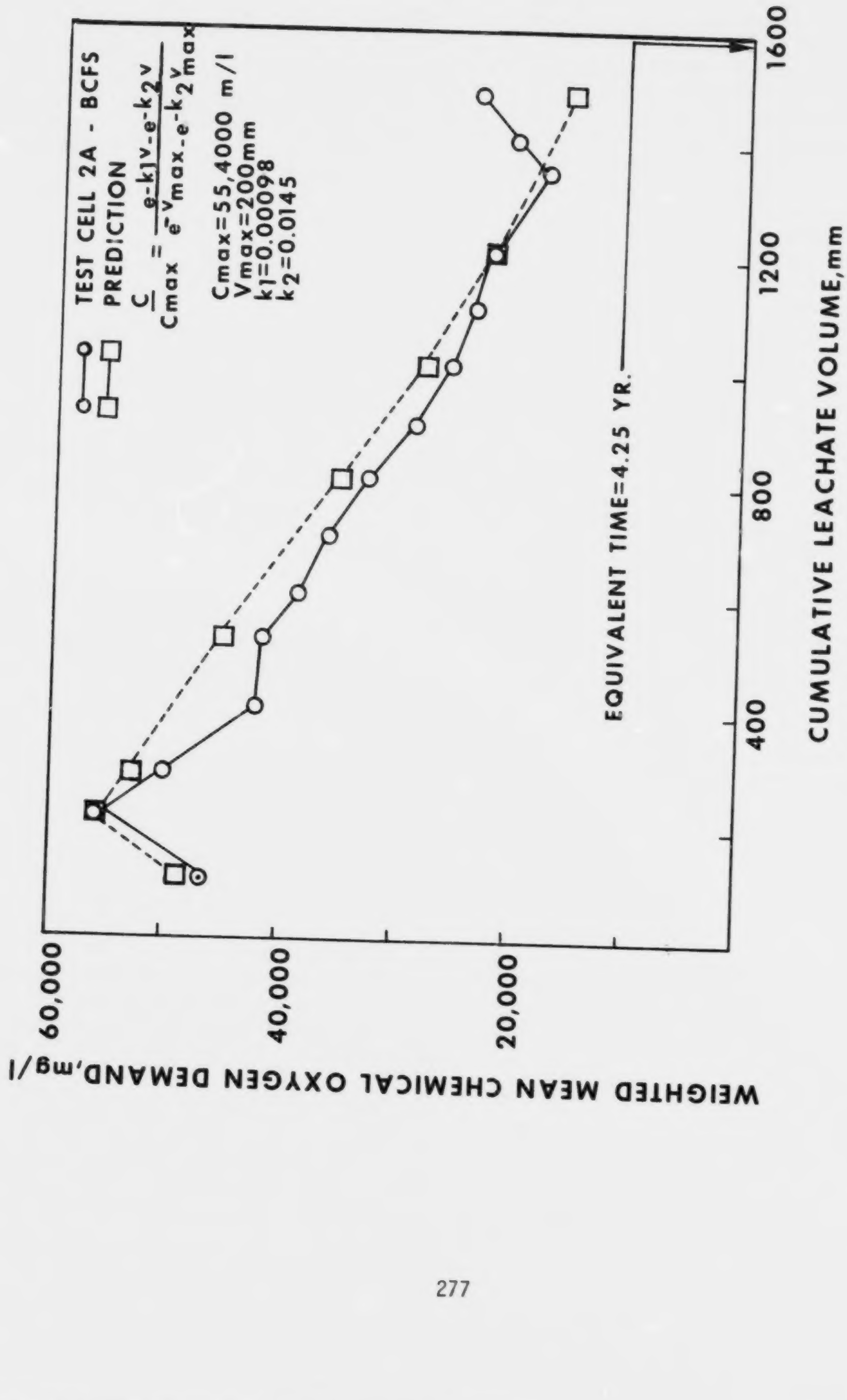

Figure 7. Empirical curve fit by wigh for Boone County Field Site Test Cell 2A.

Qasim (17) developed a concept proposed earlier (18) which assumed chloride can be used as a surrogate parameter of landfill leachate and that chloride concentration may be determined on the basis that adsorption occurs at a rate proportional to the concentration of solute in the liquid phase and the difference between the actual and maximum possible concentration of the solute absorbed on the solid particles. Comparison of empirical data with theoretical estimates were generally good during the initial phase of the study, but tended to underestimate chloride concentrations as leaching continued. Procedures for applying the concept were also provided. Relationships of dimensionless parameters and equation constants need to be verified for conditions prevalent in the field, and the long-term accuracy of estimates improved.

The limited success of initial attempts to describe leachate composition and the consistent trends of data obtained from experimental landfills indicates that further development is warranted.

Need for a Forecasting Procedure

The ability to describe leachate characteristics is considered a significant improvement in the design and regulatory review processes and in guiding the operational aspects of the landfill. It is also expected to improve the confidence and reliability of estimates describing the capability of the hydrogeological setting of any specific location to prevent contamination of groundwater. Evaluation of the continued threat to the groundwater posed by abandoned, completed, or active disposal facilities would also be possible with such a tool.

Decisions regarding the suitability of the hydrogeological setting to alleviate any threat of groundwater contamination can be improved by using a forecasting procedure to estimate the rate, duration, total quantities, and concentration of materials which will be leached. Attenuative capabilities of soils are dependent in part on the total amount of material to be removed, and the concentration. A site considered marginal due to limited attenuative capability could be utilized by identifying the amount and time during which materials should be removed from the leachate (by treatment, recirculation, or other techniques) prior to discharge to the underlying soils (Figure 8).

In a similar manner, existing disposal facilities may be evaluated to determine if corrective action is warranted to eliminate perceived or existing groundwater pollution problems. A conceptual framework is provided by a forecasting model to estimate the release pattern of pollutants. The need for corrective action can be established after field verification of the prediction and appropriate control measures selected based on estimated future contaminant release. Costly and unneeded installation of corrective measures can be avoided in a rational and environmentally sound manner.

The performance requirement of both leachate collection and treatment systems could be tailored to meet the needs of the hydrogeological setting by using a leachate forecasting procedure. Cost effective and environmentally sound decisions regarding the allowable transmission of leachate through a liner system to underlying soils, and the performance life of a liner system could be determined. Estimation of long-term costs and decisions regarding responsibility for operation and maintenance of a leachate treatment system would be aided.

Operational aspects of a landfill could be managed to affect the infiltration rates and hence the time during which the more contaminated leachates may be expected. It may be advantageous from a cost and management basis and ultimately environmentally sound to construct the landfill in a manner which will introduce greater quantities of infiltrating water during the operational life than to prolong the release of contaminants until after the site is not operating and resources may be difficult to secure for construction and operation of a control system.

The inability to describe or forecast leachate production was evident at two landfills where leachate treatment systems were designed, based on field sampling and laboratory analysis, and built but required significant modification because the characteristics of the leachates had changed. In one case, the leachate had decreased in quality to such an extent that the treatment process selected (anaerobic filter) was ineffective. The leachate composition at the other landfill had

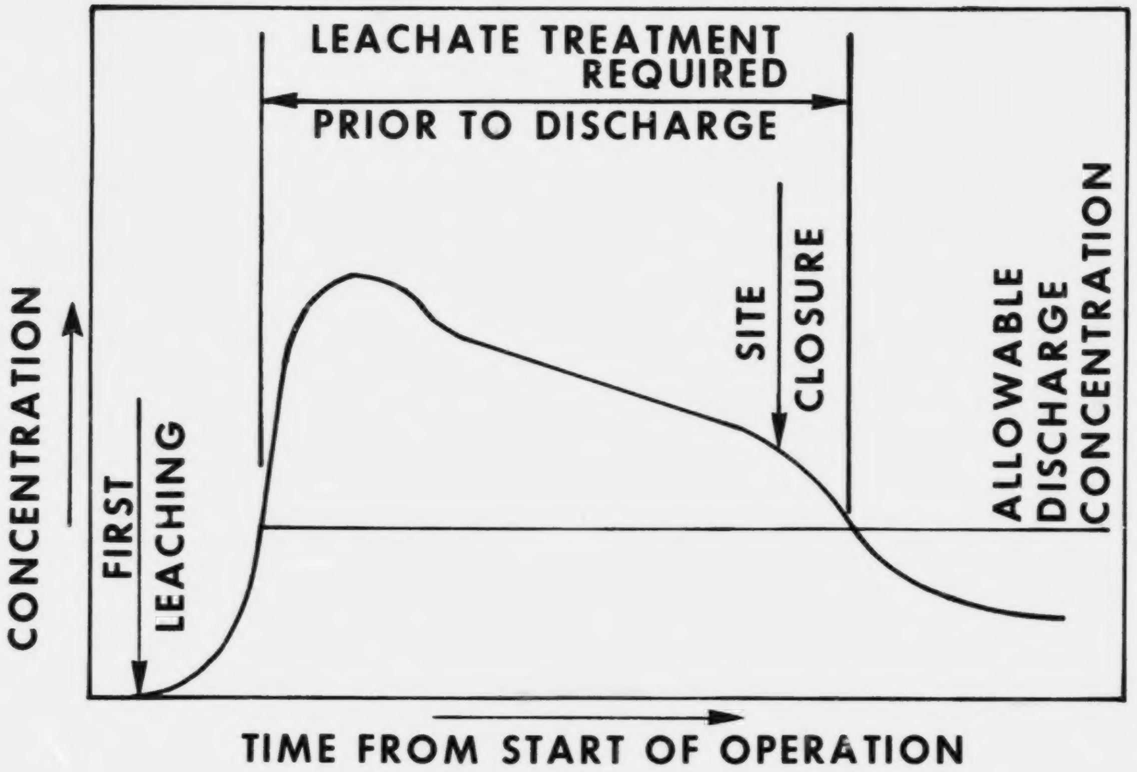

FIGURE 8. Leachate production from a refuse landfill operated for several years.

increased significantly, and additional treatment facilities were needed.

Elements of a Forecasting Procedure

A leachate forecasting procedure should be sensitive (responsive to changes) to the following items to be useful in meeting the design, regulatory and operational needs:

1. Mass of waste
2. Geometric configuration (depth, surface area)
3. Waste composition (to the extent that increasing quantities of unique wastes such as industrial or municipal sludges can be evaluated
4. Infiltration (leachate production, moisture throughput)
5. Temperature
6. Microbial activity

The list is not intended to be complete, but it is representative of conclusions from laboratory studies and field observations. The need to include the majority of these items is obvious; a special note regarding microbial activity is warranted.

The decomposition of municipal refuse proceeds through poorly-defined processes which include physical, chemical, and biological action. The role of microbial activity is expected to be very significant in determining the characteristics of leachate. Farquhar and Rovers (19) proposed several sequential phases (characterized by landfill gas composition) of microbial activity. Leachate and gas are the two modes of transport for microbial metabolic processes, consequently, significant changes in offgases may well be expected to represent microbial population and/or metabolic shifts which will also affect leachate production. The work of Pohland (15) clearly shows the interrelationship of gas and leachate composition; they are the result of dynamic microbial activity. There is, unfortunately, a severe lack of quantitative information on gas and leachate production under controlled laboratory or pilotscale conditions; explicit description of the relationship between gas and leachate production is a desirable goal, but one which will require several years to achieve with any empirical support.

The forecasting procedure should have another desirable attribute: a sound conceptual bases. The use of an empirically derived regression equation may be useful under certain conditions, but the utility of the forecasting procedure would be severely curtailed if it were not responsive to changing characteristics of landfill design and operation and solid waste characertistics. Although a myriad of physical, chemical, and biological reactions are involved in refuse decomposition, the description of the processes in an equilibrium or dynamic manner would not be possible with the existing data base; the mechanics of using the resulting forecasting procedure would likely be so burdensome as to preclude its use.

Identification and development of the primary phenomena controlling leachate production would be a preferable basis to such a complex approach. For example, leachate composition may be controlled by rate limiting reactions, where the composition is principally dependent on the rate of leachate movement through the waste because the rate of solubilization is slow relative to the rate of leachate flow. Saturation limiting reactions may also provide a better description, where composition is dependent on complex solution equilibria which occur fast relative to leachate flow. Concentration gradients between the flowing leachate and a stagnant water film over the refuse particles is being evaluated as a basis for a forecasting procedure (20). Certainly, the composition of leachate will be dependent on the finite material available for solution. Ham and Stegman (21) have described phenomena which were considered important in their developmental work on waste extraction procedures. A review was also provided in a compilation of extraction and leaching procedures (22).

Development of a useful forecasting procedure must be responsive to incremental additions of solid waste as is encountered in day to day landfill operations. The proposed procedure must also be verified by comparing projected composition and mass removals with long-term observations made at field sites where volume and composition of leachate, the mass of refuse and other parameters are measured.

Summary

An appreciable amount of information on leachate production has been collected over several decades. Results of several experiments seem to follow very consistent trends which can be described using empirical models, but these models are limited in their applications, because they describe leachate from batch-type laboratory and pilot scale experiments of relatively shallow depth. A need exists for a leachate forecasting procedure which will be sensitive to waste characteristics, site geometry, infiltration, temperature, and microbial activity. Such a procedure would be useful for design, operation and regulatory purposes. A conceptual framework would be established from which quantitative and rational estimates of leachate production, could be made. These estimates are needed for designing, and operating leachate collection and treatment systems, for assessing the adequacy of the site hydrogeology to prevent groundwater contamination, for planning the operational filling of the disposal site, and for assessing the need for remedial or corrective action at completed or operating disposal facilities to alleviate a real or suspected groundwater contamination condition. The procedure should be conceptually sound but should not be so complex as to inhibit its usefulness.

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PILOT-SCALE INVESTIGATIONS OF ACCELERATED LANDFILL STABILIZATION WITH LEACHATE RECYCLE

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ABSTRACT

Pilot-scale investigations of accelerated landfill stabilization with leachate recycle continue to demonstrate the advantage of this solid waste management option with respect to process control and predictability. Stabilization of readily available organic materials contained in shredded municipal solid waste and transferred to recycled leachate was essentially completed within about six months with a concomitant accelerated production of gas. Landfill behavior during periods of rapid stabilization followed predictable patterns as measured by selected parameters including TOC, COD, BOD_5, total volatile acids, pH, conductivity, total alkalinity, gas production and composition, ORP, and chlorides. The inter-relationships of these parameters and their application for interpretation of landfill stabilization during leachate recycle are described within a landfill management perspective.

INTRODUCTION

Emerging regulations governing land disposal of solid wastes continue to require safeguards against the migration of reaction products from landfill sites into the environment. Much of this emphasis arises due to the concern over the consequences of leachate and gas production. The production of both leachate and gas are inextricably related and therefore tend to manifest themselves in an often unpredictable and elusive fashion. Under such circumstances, the environmental impacts of leachate and gas migration from landfills are frequently evaluated only after the fact and without much opportunity for providing an immediate and lasting remedy. This dilemma is a direct outgrowth of past landfill management practices and a tradition of attempting to isolate all landfill sites and store their contents in perpetuity.

From an engineering point of view, it is most important to ensure and maintain operational control over landfill disposal of solid wastes in order to provide lasting guarantees against environmental impairment. Therefore, the concept of confinement of the site and control over the reaction products within the site becomes an attractive option. Indeed, with the enactment of the Resource Conservation and Recovery Act of 1976 (P.L. 94-580) and the resultant guidelines for the control of hazardous wastes, such a management strategy almost becomes a necessity. Confinement, however, implies isolation, accumulation, collection, and regulated treatment or release of contaminants for which several procedures may be considered.

Recognizing the eventual need for collection and treatment of leachate at certain landfill sites, efforts have been made to investigate the relative propriety of both biological and physical-chemical treatment methods. These have been presented and reviewed elsewhere(1-3) with descriptions of operational characteristics, limitations, and relative economics of application. Collectively, these efforts indicate that preparations for separate leachate treatment, except possibly for controlled discharge to an existing sewerage system, become exceedingly challenging particularly in view of the many uncertainties regarding leachate quantity and quality. Moreover, since actual leachate production, without intentional addition of moisture to the landfill either during operation or after closure, is generally delayed for an indeterminate number of years, treatment becomes an ex post facto proposition, the eventual implementation of which and its effectiveness then becomes a matter of considerable conjecture.

To alleviate some of the problems heretofore described and to address the issues of control and predictability of landfill behavior, research on the concept of leachate containment, collection and recycle utilizing the landfill as a treatment system has been conducted at Georgia Tech for the past eight years. It is the purpose of this report to review selected results of more recent studies and to demonstrate that leachate recycle is a viable management option for solid waste disposal sites.

EXPERIMENTAL TECHNIQUES

Prototype Landfill Configuration

To augment the results of the initial landfill column studies(1) and to permit investigations on a larger scale, leachate recycle studies were extended with the construction of two simulated landfill cells as illustrated in Figure 1.

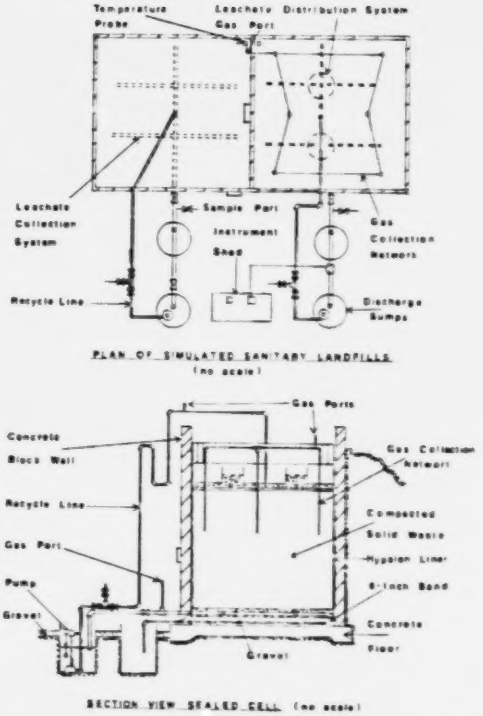
Temperature Probe
Leachate Distribution System
Gas Port
Leachate Collection System
Sample Port
Instrument Shed
Gas Collection Network
Recycle Line
Discharge Sumps
PLAN OF SIMULATED SANITARY LANDFILLS
(no scale)
Gas Ports
Concrete Block Wall
Gas Collection Network
Recycle Line
Compacted Solid Waste
Gas Port
Hypalon Liner
Pump
6-Inch Sand
Gravel
Concrete Floor
Gravel
SECTION VIEW SEALED CELL (no scale)

Figure 1. Pilot-Scale Landfill Cells with Leachate Recycle

The essential differences between the operation of these cells and the original columns were that the new cells were filled with about ten feet of shredded municipal solid wastes, were allowed to reach field capacity in response to natural rainfall, and were constructed to permit gas collection and analysis of the effects of evaporation. One cell was left open to incident rainfall, the other was sealed to allow for gas measurements and to prohibit loss of moisture by evaporation. Accordingly, the amount of rainfall received by the open cell was measured and an equivalent amount of tap water was added to the sealed cell.

The landfill cells were each constructed of reinforced concrete, 10 feet square by 14 feet deep, with a sloping base. All exposed surfaces were covered with a sealant and the interior-walls were cushioned with styrofoam prior to insertion of a continuous Hypalon ^{R} liner. This liner was left open at the top in the open cell and covered to form a completely sealed enclosure for the sealed cell. This enclosure was finally sealed with a 0.25-inch thick steel lid.

A system of underdrains bedded in gravel was placed both above and below the liner in the bottom of each cell to permit leachate collection for recycle and examination for liner integrity with time. Leachate collection was accomplished with 55-gal sumps from which it could be pumped through a distribution system placed in gravel at the surface of each cell just above the solid waste and below a 2-foot layer of soil cover.

The conduits for leachate collection and recycle were of 2-inch PVC pipe; underdrains and distribution laterals were perforated with 0.5-inch diameter holes drilled 4.0 inches on center. Two gas collection networks were also employed with the sealed cell. The principal gas collection system consisted of a length of 0.75-inch diameter PVC pipe extending from the leachate distribution conduit. The alternate system was designed to collect gas which had migrated into the void space above the cover soil and below the confining lid. In addition to these gas collection systems, temperature probes were placed into the interior of each cell to provide a continuing record and comparison of cell temperatures with ambient conditions.

Facility Operation

Upon completion of the construction of the landfill containment structures, each received residential-type solid waste collected and shredded to a nominal size of 2.5 to 3.0 inches in DeKalb County, Georgia. The shredded solid waste was delivered to the facility site and placed in 2.0-foot increments to a manually compacted density of 537 pounds/cu.yd. The final compacted depths of solid waste were 9.0 feet and 8.5 feet in the open and sealed cells, respectively. The initial filling operation was completed on August 13, 1976 with the installation of the leachate distribution system, the final 2.0 feet of cover soil, and closure of the sealed cell. Over the next year, the cells were allowed to reach field capacity with incident rainfall or water addition to the respective open and sealed cells. During this period, rainfall, temperature and relative humidity were recorded, the cells were checked for any leachate production, and the sealed cell was sampled for possible gas evolution.

Although some leachate appeared in both cells in small quantity on an intermittent basis, it did not appear regularly until the Fall of 1977. Accordingly, September 1, 1977 was selected as the time that leachate production nominally began. Thereafter, samples were periodically collected for analysis from the recirculation sump of each cell and available leachate was recycled at about weekly intervals. By day 208, enough leachate was available to initiate daily recycle and 200 gallons/day/cell were recirculated. This procedure was employed until day 346 when the daily recycle quantity was reduced to that quantity capable of being passed through the open cell in one day. Every other day recycle was initiated on day 401 and the amount recycled through each cell was gradually reduced to about 40 to 50 gallons.

Analytical Procedures

As the solid waste was being placed, six compacted samples were collected and analyzed for density, moisture, total volatile solids, calorific value, and CHN content. The leachate samples collected from both cells were analyzed for pH, ORP, conductivity, volatile acids, TOC and

TIC, BOD_5, COD, total suspended, and volatile solids, total alkalinity, nitrogen, phosphorous, sulfates, Chlorides and selected metals (Fe, Ni, Mn, Cr, Cd, Mg, Cu, Na, K, Ca, Zn, Al, Pb). Gas samples were analyzed for CO_2, O_2, N_2, CH_4 and H_2. Standard Methods(4) techniques, supplemented with instrumental analyses, were used thoughout the studies.

EXPERIMENTAL RESULTS AND DISCUSSION

Selected data from the studies are presented in Table 1 and Figures 2 through 13. These data reflect the most pertinent results of environmental conditions existing during the test period as well as solid waste characteristics and analyses on leachate and gas samples. The time scales used in this presentation, i.e., time since placement of solid waste and time since leachate production began, are related in that nominal leachate production began 383 days after the solid waste was initially placed in the cells.

Solid Waste Characteristics

The original character of the solid waste utilized in the study is included in Table 1. These analyses were intended to provide some comparative information on the constituent nature of the solid waste and an opportunity for final characterization after leachate recycle and stabilization had been completed.

Environmental Conditions

The weekly ranges in maximum and minimum ambient and internal cell temperatures are presented in Figures 2 and 3, respectively. These temperatures were continuously monitored, and although diurnal variations were observed, weekly averages of low and high temperatures were considered sufficient to display the seasonal variations to which the cells were exposed. Accordingly, ambient temperatures varied from a low of -12.2°C in January 1977 to a high of 42.2°C in July 1977. Because of some insulation, these variations in low and high temperatures were not so dramatic in the landfill cells. The open cell ranged from a low of 12°C to a high of 38.5°C and the sealed cell varied between a low of 6.5°C and a high of 41°C. The somewhat lower and higher temperature ranges in the closed cell were considered a consequence of the isolated nature of that cell which influenced the effects of warming during the day and cooling at night. However, the differences noted between the average temperatures of the two cells were not considered significant to the overall stabilization rates of the solid waste and leachate between the two cells at least within the time frame of the study period.

The incremental and cumulative rainfall incident on the open cell and added as tap water equivalent to the sealed cell are indicated in Figure 4. Except for some relatively dry periods during the

Table 1

Original Characteristics of Solid Waste Used During Pilot Scale Investigations with Leachate Recycle

Parameter	Average Analysis*
Density as placed, lb/yd^3	537
Moisture Content as placed, %	33.5
Total Volatile Solids, % dry	75.5
Carbon, % dry	45.3
Hydrogen, % dry	5.46
Nitrogen, % dry	3.33
Calorific Value, BTU/lb dry	7758

*Average of six representative samples.

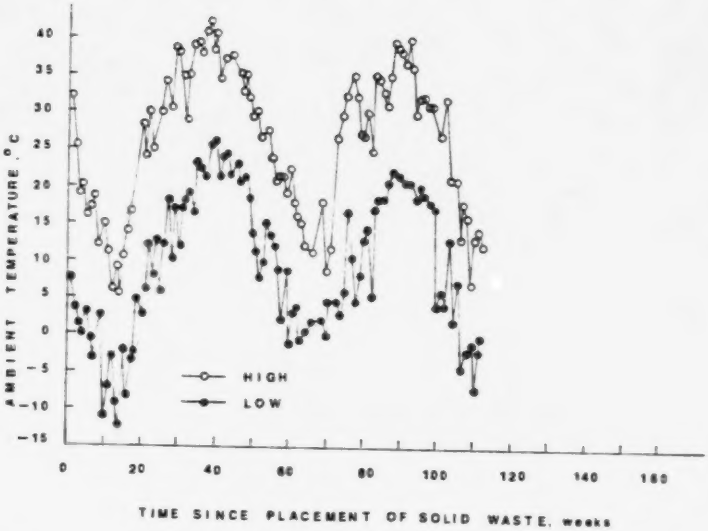

Figure 2. Ranges of Weekly Maximum and Minimum Ambient Temperatures at Test Site

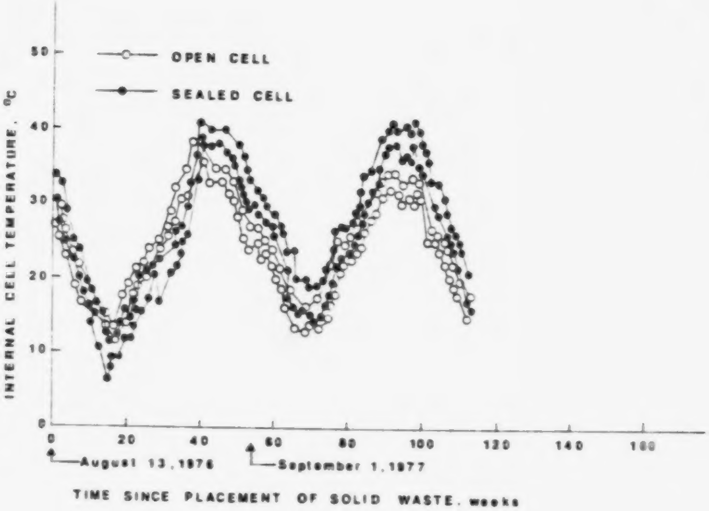

Figure 3. Ranges of Weekly Maximum and Minimum Temperatures within the Landfill Cells

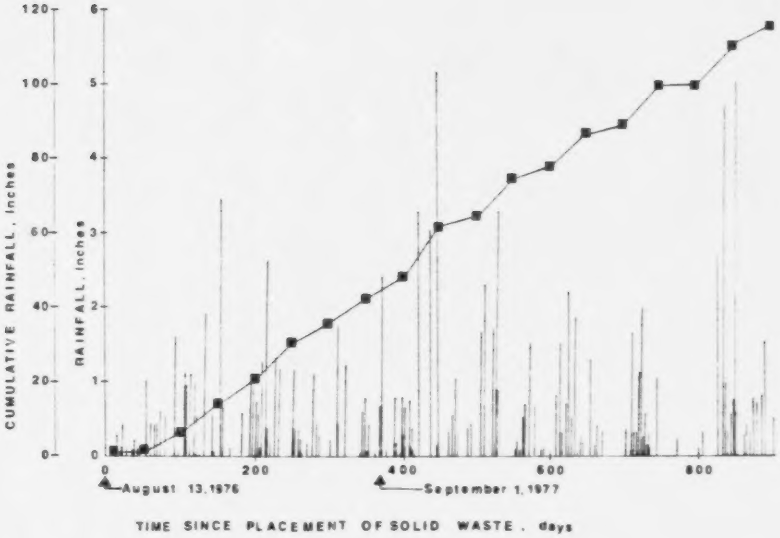

Figure 4. Incremental and Cumulative Rainfall at Test Site

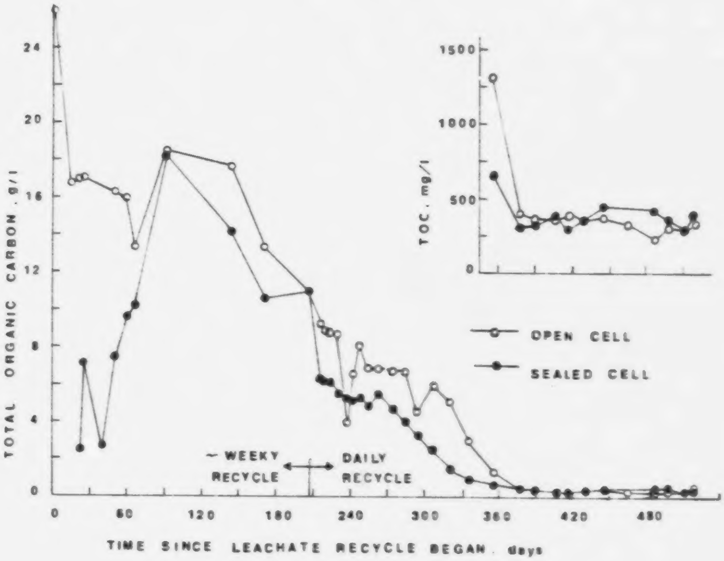

Figure 5. Total Organic Carbon Concentrations in Leachates from Pilot-Scale Landfill Cells

latter stages of the study, rainfall was relatively consistent with an accumulation of about 120 inches at the end of the report period.

Leachate and Gas Characteristics

Selected analytical data accumulated during 520 days of leachate production with either intermittent, daily or every other day leachate recycle are indicated in Figures 5 through 13. Recognizing that those parameters commonly employed to measure pollutional potential are particularly germane to the identification and solution of problems associated with the production of leachate, TOC, COD and BOD_5 were used to reflect that pollutional potential derived from the decomposition of organic matter. As indicated in Figures 5, 6 and 7, all three parameters exhibited essentially the same pattern with leachate recycle. During initial weekly recycle periods, the data were somewhat erratic particularly for the sealed cell where the even distribution of rainfall across the entire surface of the landfill apparently was not simulated by the introduction of equivalent water through the recycle distribution system of the sealed cell. However, once daily recycle procedures were initiated, these variations were moderated in both cells and more even and rapid decreases in pollutant concentrations were experienced.

Notwithstanding dilution effects, the rapid decline in pollutant concentrations in the leachate as consequenced by daily recycle was considered primarily indicative of an initial acceleration of biological stabilization of the more readily available organics contained within each cell. Consequently, daily recirculation of leachate served to provide a continuing exposure of the internal biological populations to nutrients contained in the leachate and thereby enhanced overall conversion of those constituents as well as those in the solid waste and transferred to the leachate during passage.

Since biological stabilization during landfill disposal of solid waste depends largely on anaerobic microbial activity, it is possible to further interpret the observed changes in pollutional characteristics of the leachate. If the two-phase process of acid fermentation with the production of intermediates such as the volatile acids followed by fermentation of these acids to CH_4 and CO_2 is applied, then the results of Figures 5 through 7 should also be reflected by an initial appearance and subsequent utilization of volatile acids. Indeed, inspection of Figure 8 indicates a similar rise in volatile acids followed by their virtual elimination (below detectable limits) particularly after daily leachate recycle was instituted. It is also interesting to note that reductions in these parameters in the sealed cell preceded those in the open cell although the eventual results were the same. This difference could be attributed to the preferred environment provided to the volatile acid utilizers by the sealed cell, i.e., a more positive exclusion of oxygen and its detrimental influence on the obligate anaerobic methane formers.

As could be anticipated, as volatile acids accumulated, pH decreased as indicated in Figure 9. Moreover, this decreased pH condition was sustained until the volatile acids were converted and removed; the change in pH toward neutral was again more rapid in the leachate of the sealed cell than in that of the open cell. This pH change in both cells was indicative of a buffer shift from that established by the volatile acids to that of the more favorable bicarbonate buffer system which is also considered necessary for efficient CH_4 production. This shift is furthermore illustrated by decreases in conductivity as well as in total alkalinity attributable to the volatile acids as illustrated in Figure 10.

Following the production and accumulation of volatile acids with the con-

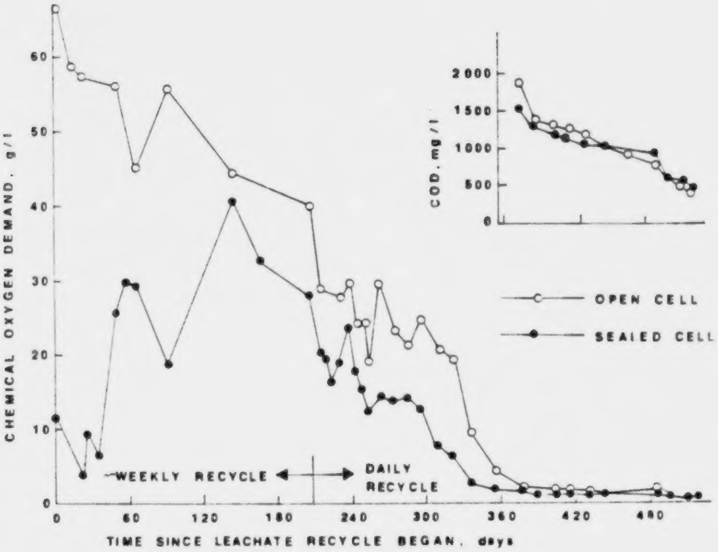

Figure 6. Chemical Oxygen Demand of Leachates from Pilot-Scale Landfill Cells

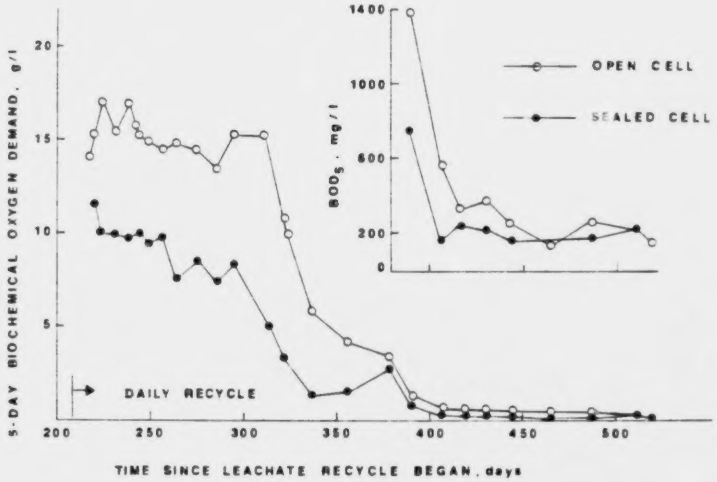

Figure 7. Biochemical Oxygen Demand of Leachates from Pilot-Scale Landfill Cells

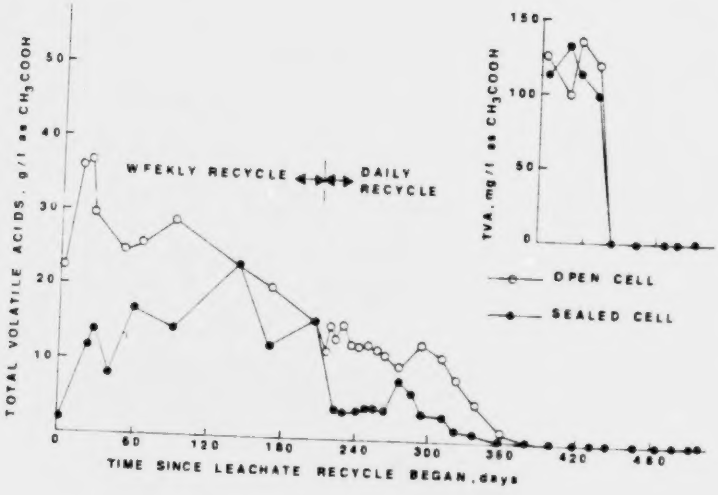

Figure 8. Total Volatile Acids Concentration in Leachates from Pilot-Scale Landfill Cells

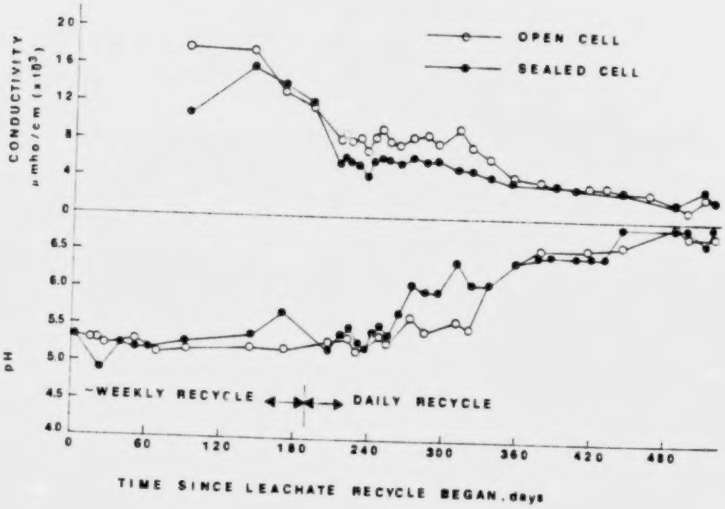

Figure 9. Conductivity and pH of Leachates from Pilot-Scale Landfill Cells

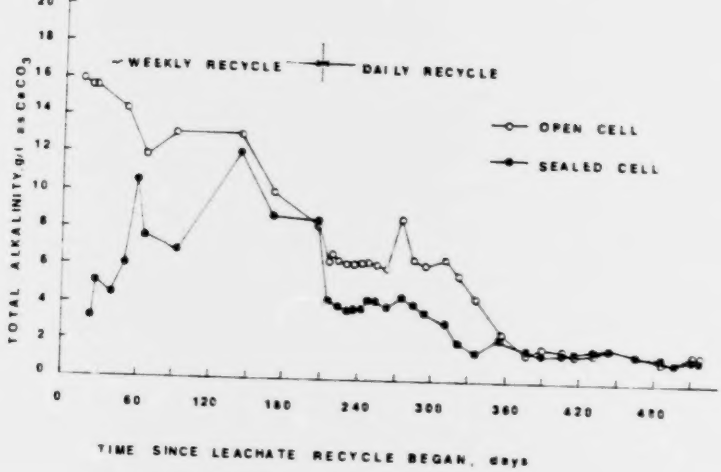

Figure 10. Total Alkalinity of Leachates from Pilot-Scale Landfill Cells

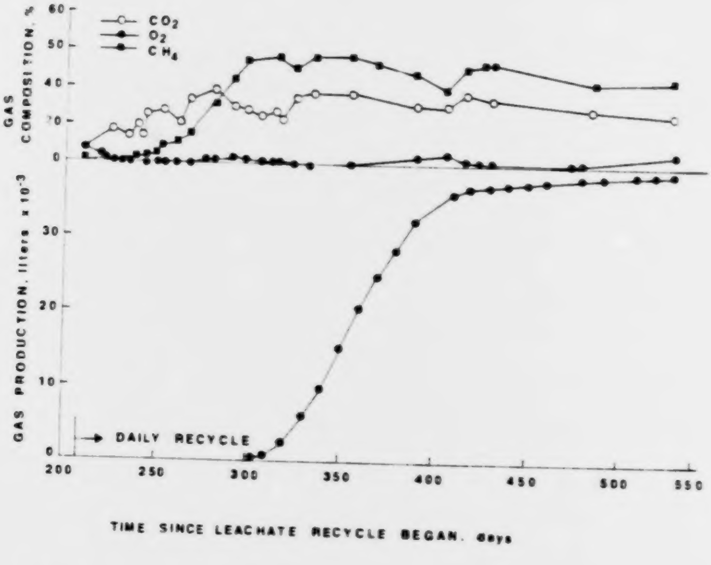

Figure 11. Gas Production and Composition from the Sealed Pilot-Scale Landfill Cell

comitant decline in pH, gas production in the sealed cell began to increase and accelerate as environmental conditions became more favorable for the growth and proliferation of the CH$_4$ formers. Accordingly, gas production increased commensurate with the reduction in volatile acids and increase in pH. Whereas, during periods of volatile acid accumulation, little methane was formed, soon thereafter, gas production accelerated to reach a high of 639 liters per day with a CH$_4$ content of 57% normalized to about 60% as indicated on Figures 11 and 12. This period of accelerated gas production coincides with the period of most rapid stabilization and although gas production was still being measured at the end of the report period, it had diminished to about 10 to 20 liters per day. Consequently, the greatest majority of gas production from the readily available organic pollutants in the leachate had been completed during the relatively short period of about three months when conditions favoring CH$_4$ production were most optimum. These optimum conditions are also reflected by ORP measurements (Figure 12) and, although not as negative as generally reported as necessary for most efficient CH$_4$ production during anaerobic biological stabilization(5), the trend is indicative of requisite negative values and probably were not more so due to sampling difficulties associated with the unavoidable exposure of samples to oxygen (air) during sampling and analysis.

Finally, since dilut [illegible] consequenced by the addition of rainfall or water equivalent to the two cells would result in some dilution of pollutant concentrations, a conservative constituent could be chosen to reflect the differences in moisture accumulation in the open cell which was reduced by evaporation opportunities and that amount accumulated in the sealed cell where evaporation was disallowed. Inspection of Figure 13 provides some indication of this difference of accumulated moisture between the two cells at the end of the report period. Accordingly, concentrations of chloride in the open cell, where some moisture was lost to evaporation, are higher than in the sealed cell where more dilution capacity is available. Collectively, these results indicate that between 20 to 30 percent of the rainfall incident on the open cell during the report period was lost due to evaporation. The relative degree of dilution afforded by that amount of rainfall thereby contributing to the accumulated quantity of leachate has important ramifications with respect to actual masses of materials contained in the leachates. For instance, the chloride concentration exposed to a clay containment liner is important to prediction of the rate of its migration into the surrounding environment particularly if it is used as a tracer and/or quality standard. In these experiments, the chloride concentration diluted by water equivalent to incident rainfall was about 325 mg/ℓ; with the indicated evaporation, this concentration increased to about 410 mg/ℓ.

SUMMARY AND CONCLUSIONS

Continuing research investigations with pilot-scale landfill cells and leachate containment, collection and recycle have supported the results of previous work establishing the advantages of such a landfill management option in promoting more rapid and predictable stablization of readily available organic constituents with concomitant increased rates of gas production. Decreasing the time for such stabilization to a matter of months rather than years provides attractive opportunities for energy recovery as well as rapid realization of potentials for land reclamation and/or ultimate use without the uncertainities attendant with present landfill management practices. The data presented herein should better enable the engineer and scientist to consider leachate recycle as one option for better design, operation and control of landfill disposal of solid waste with respect to practicability, economics and safeguards against environmental impairment.

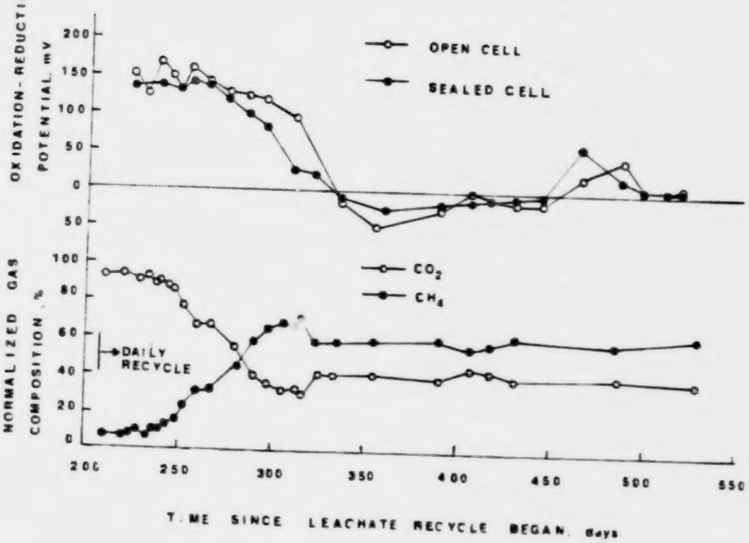

Figure 12. Normalized Gas Production and Oxidation-Reduction Potential of Leachate from the Sealed Pilot-Scale Landfill Cell

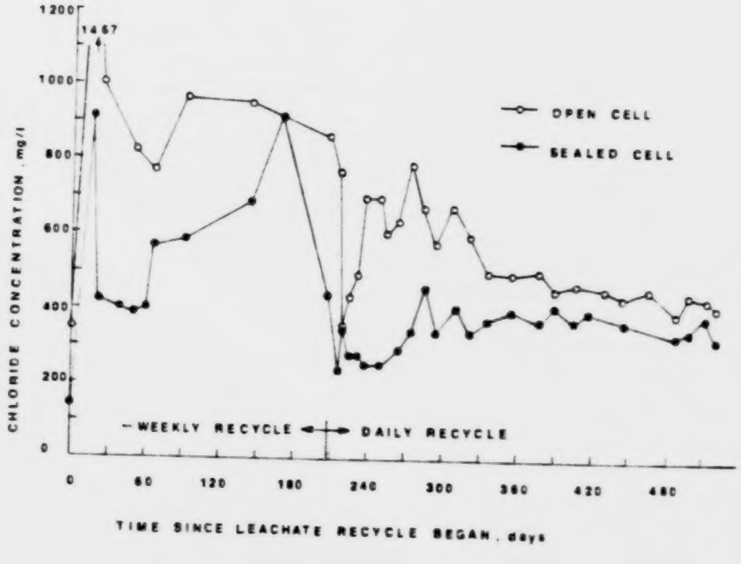

Figure 13. Chloride Concentrations in Leachates from Pilot-Scale Landfill Cells

ACKNOWLEDGEMENT

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LEACHATE TREATMENT SCHEMES - RESEARCH APPROACH

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ABSTRACT

Leachates are produced when surface and/or groundwater pass through layers of refuse in a landfill site and become contaminated. If these leachates enter a nearby receiving water, a serious water quality problem may result, with potential for fish kills. It is now generally accepted (and law in some areas) that landfill leachates and their effects be controlled, primarily through collection and treatment of such.

The purpose of this paper is to provide a brief overview of the approach to and research results, to date, of a comprehensive long-term, solid waste disposal/leachate treatability study; this project has been underway at the University of British Columbia, since 1971, and is under the direction of personnel in the Environmental Engineering Group of the Department of Civil Engineering. The approach to variable-strength, leachate treatment, brief summaries of several of the research projects completed to date (or about to be) and preliminary information on scale-up to on-site lagoon treatment of an insitu leachate, are presented in this manuscript. Topics requiring further investigations are also outlined.

INTRODUCTION

Sanitary landfills still remain the most popular method of solid waste disposal. However, one of the major problems presented by landfills, particularly in high precipitation climates, such as the Pacific Northwest, is the production of leachate. These leachates are produced as surface and/or groundwater pass through layers of refuse in a landfill site and become contaminated. If this leachate enters a nearly receiving water, a serious pollution problem may result, with a potential for fish kills. The magnitude of such pollution depends on a number of factors, including strength and quantity of leachate produced, as well as the dilution being afforded by the receiving water. The former, in turn, depend on a number of variables, including composition of the refuse, age and hydrogeology of the site, and of course, the climate.

Various adverse environmental effects due to leachates have been well documented in the literature (1,2,3). Although leachate composition varies widely, it is often classified as a high strength wastewater, that is, the levels of such wastewater parameters as BOD, COD, suspended solids, and turbidity are many times greater than those found in municipal wastewater. Additional characteristics such as low dissolved oxygen, low pH, and a spectrum of toxic chemicals and metallic ions are quite commonplace. Table 1 illustrates the observed variability of leachate strength and composition (1).

It is now generally accepted (and law, in some areas) that landfill leachates and their effects be controlled primarily through collection and treatment of such. This concept is being applied to both existing fills and proposed sites. Treatment of leachates might include on-site schemes, discharge to an existing municipal sewer line (with downstream combined treatment) or even separate on-site storage and periodic hauling to the nearest treatment plant.

A number of factors are involved in this decision making process, including of course, proven treatment technology for a leachate-type waste. The latter, in turn, is currently the subject of extensive on-going research in the water pollution control field. This paper is an attempt to summarize the approach and results (to date) of part of a long-term solid waste disposal/leachate treatability research study; this project has been underway since 1971 at the University of British Columbia, under the direction of personnel in the Environmental Engineering Group. The initiator and overall coordinator of this project is Dr. R.D. Cameron of the Department of Civil Engineering.

BRIEF PROJECT HISTORY AND RATIONALE

(i) Solid Waste Disposal Study

As indicated above, this study is a long-term one and is now in its eighth year of operation. It is solely under the direction of R.D. Cameron and only remotely involves this author. Although the number of personnel associated with this particular project has fluctuated over the years, currently, the support staff includes four full-time technicians, one full-time and one half-time Research Associates. In addition, this project is backed up by the finest available instrumentation.

The purpose of this on-going program is to characterize landfill leachates and monitor variations in their composition with time, precipitation rate, cover material and other parameters. This study is being carried out on existing landfills and a battery of special lysimeters. A total of thirty-nine various sized lysimeters are currently in operation, the largest of which are outlined in Table 2. A major part of this study involves thorough examination of co-disposal of industrial and municipal wastes, including so-called "toxic and hazardous" compounds. Extensive toxicity studies are also being carried out. Interested readers are urged to contact the principle investigator directly for further details and information on this aspect of the project.

(ii) Approach to Leachate Treatability

The Pacific Northwest, including the province of British Columbia, is quite famous for its high levels of precipitation. This is particularly true for the City of Vancouver and the surrounding areas. This area is also quite famous for its fishing industry, in particular the commercial salmon fishing industry. The latter is worth millions of dollars annually to both Canadian and U.S. fishermen.

Considering these two facts and the extensive use of landfills for disposal of solid wastes, one quickly realizes that a major problem exists - the production of leachates of varying strengths and composition and its potential for serious receiving water pollution and fish kill. Because of today's general environmental awareness and concern, as well as stricter pollution control laws, governing both discharge and receiving water quality, the collection and treatment of leachates throughout the province is now an accepted fact.

Because of the nature and complexity of the problem and the lack, in literature, of convincing and proven treatment technology for such an unusual waste (especially high-strength versions), the research approach undertaken by this author and his colleagues involves several comprehensive and deliberate steps. The approach to developing viable treatment methodology is being carried out as follows:

(a) The leachate sources are both lysimeters as described above, and existing landfill sites in the Vancouver area. In this manner, a cross-section of leachate composition and strength can be obtained, with a goal toward simulating the progression of landfills from young to old sites. In addition, treating the leachate "as is", whether exceptionally strong or weak, injects a distinct note of realism to the project, i.e. field conditions.

(b) Examination of a complete cross-section of possible treatment schemes, although formidable in concept, is imperative. In a sense, a "shot-gun" approach has been used with a goal toward narrowing down the choice(s). The latter, in turn, would be based on proven effectiveness, viability and cost-effectiveness, in conjunction with on-site treatment or combined municipal-leachate treatment. The scope of research projects includes aerobic-biological, anaerobic-biological, chemical/physio-chemical, joint biological-chemical, and other more exotic treatment schemes.

(c) These research projects are being undertaken at the laboratory or bench-scale level for the time being. Once the treatment effectiveness has been proven at this level and the field of choice, so to speak, has been narrowed down, scale-up to pilot-plant or full-scale will be examined in conjunction with (d) below.

(d) Pending successful bench-scale treatment of any one particular leachate, optimization is planned. This is to be carried out in connection with a demonstration pilot-plant or integrated treatment scheme in the field, should time and/or funding be a critical factor. Under extended circumstances, optimization and prediction for scale-up are to be performed in the laboratory.

(e) The project scope not only includes the personnel within the Environmental Engineering Group at the University, but invites municipal, provincial and federal participation and working assistance. The concept of "team approach", in an on-going study, is of vital concern to the momentum of this program.

Considering the aforementioned points, the following section provides a synopsis of some of the "leachate treatment schemes" examined to date. In some instances, the projects have been completed and the results fully or partially documented in the literature; in other cases, the work has just been completed, or is about to be, and only preliminary results are presented.

SUMMARY OF TREATMENT SCHEMES

(i) Project Title: Physio-Chemical Treatment of a High-Strength, Sanitary Landfill Leachate

Summary

Preliminary results from this project have been presented elsewhere (4). The entire study is now in the final write-up stage and further results will shortly be available.

The research described here-in was aimed at developing a physical-chemical treatment process for a strong, landfill leachate, such that the effluent might lend itself to controlled discharge to a biological treatment system or, if necessary, to a natural receiving water. The lysimeter-generated leachate composition is shown in Table 3.

Preliminary screening, using a Placket-Burman two-level, fractional factorial design (4), indicated that only lime, alum, ferric sulphate, ozone and synthetic polymers, individually and in combinations might be significant treatment reagents, out of a possible 12 variables tested; however, subsequent work demonstrated only lime and ozone to be significantly effective in treating the raw leachate. Ozone was applied via a specially designed, contacting system (described elsewhere (4)) while all chemical and physical reagents, including lime, were tested in conventional jar-testing apparati.

The results essentially indicated that lime was quite effective in removing many of the monitored pollutants, with "best treatment" dosages at approximately 2350 mg/L. Ozone was mainly effective only after 100 mg/L had been applied. The leachate was found to have a "constituent ordered" ozone demand that must be satisfied before any other pollutant is significantly removed. This demand was primarily due to the dissolved organic content of the waste. Best treatment was considered in the context of cost-effectiveness, not necessarily maximum applied dosages.

The combination of lime and ozone, in specific concentration ranges, was found to effectively remove or reduce many of the metallic ions, adjust pH, and even disinfect this high-strength leachate. Colour and turbidity were dramatically reduced - however, organic carbon and COD reduction were limited, essentially peaking at 48% and 34% respectively, in the context of "best treatment". It was also recognized that the sludge produced in this physical-chemical treatment process could pose additional problems, both in treatment and disposal.

(ii) Project Title: Treatment of a Complex Landfill Leachate With Peat

Summary

All aspects of this research project have been completed. The final results have been published elsewhere (5). The basis for this project was previous work by Lidkea (6) and Tinh et al. (7).

The aim of this study was to use cheap, locally-available peat as a treatment scheme for landfill leachates, since some of British Columbia's largest landfills (i.e. Vancouver's Burns Bog Landfill) are located on peat bogs. The leachate tested was collected from this local landfill site and the composition is shown in Table 4.

This study incorporated the passage of leachate through plexiglass columns filled with an amorphous-granular peat. Preliminary adjustment of pH showed that reducing the pH downward to 4.8 dramatically reduced the metal adsorption capacity; however, upon increasing the pH to 8.4, the metal removal was significantly increased, owing to the filtration of precipitated metals. The best adsorption of metals seemed to occur at the "natural" pH of 7.1.

In this work, manganese was found to be the limiting pollutant. At the 0.05 mg/L maximum acceptable concentration allowed by the British Columbia Pollution Control Branch (8), 94% of the total metals were removed, thus requiring 159 kg (350 lb) of peat per 1000 litres of leachate. Resting the peat for 1 month did not significantly increase the removal capacity.

This research project also demonstrated that desorption of some contaminants occurred when water was percolated through the peat. The desorption-test effluent was not toxic to fish, but iron, lead and COD exceeded acceptable values (8). Chemical pre-treatment of the leachate, using lime and ferric chloride, did achieve significant iron, manganese and calcium removals; however, chemical pre-treatment, followed by peat adsorption, offered no advantages, other than reducing toxicity to fish. Therefore, peat treatment alone was quite effective in reducing leachate contaminant concentrations to a level that was non-toxic to fish (rainbow trout). Table 5 summarizes the overall findings of this study (5).

(iii) Project Title: Treatability of Leachate From a Sanitary Landfill By Anaerobic Digestion

Summary

This research project was successfully completed in 1974 and the final results have been published elsewhere (9). A synopsis is presented herein.

The study investigated the possibility of reducing large amounts of oxygen demanding material in a high-strength leachate through the process of anaerobic digestion, without any prior removal of heavy metals. This work also included a study of the effects of varying detention times on process performance and leachate characteristics. The digesters employed were of 14 litres capacity and were operated at 35°C.

The strength of the leachate was such that influent BOD_5 values ranged from 11,000 - 16,000 mg/L - nevertheless, BOD_5 removals ranged from 80 to 96 percent over detention times of 5 to 20 days. Similarly COD removals ranged from 65 to 79 percent for influent values of 23,000 to 33,000 mg/L.

The leachate feed also contained a variety of metals, including aluminum, cadmium, chromium, copper, lead, mercury, nickel and zinc - zinc exhibited the highest influent concentration at 65 mg/L. Despite the presence of these metals, the anaerobic digestion process seemed to function consistently. Many of these metals associated themselves almost completely with the sludge portion of the effluent.

Gas production, in this study, was monitored on the basis of BOD and COD destroyed - 11.9 to 15.0 cubic feet of total gas was produced per pound of BOD_5 destroyed. This digester gas contained 68 to 76 percent methane, with the percentage decreasing with increased detention time. On the basis of COD, 5.8 to 6.8 cubic feet of methane was produced per pound of COD destroyed. (Note: cu ft/lb x 0.0624 = m^3/kg).

Despite reasonably good treatment efficiency at detention times up to 20 days, residual BOD_5 values ranged from 400 to 3000 mg/L - thus, further treatment would obviously be mandatory to meet appropriate discharge levels. Settling the effluent for one half hour reduced these residuals by 13 to 57 percent.

(iv) Project Title - Aerobic Biostabilization of a High-Strength, Landfill Leachate

Summary

This project was completed in 1976 and was the first in an on-going series of aerobic, biotreatment studies. The results have been widely published and comprehensive data can be found elsewhere (10,11,12) - only a brief overview is presented herein.

This study concerned itself with treating a high-strength leachate (lysimeter generated) by aerobic, biological methods, without any prior removal of the heavy metals or toxic substances contained in that leachate. The effect of varying sludge age was also investigated and the distribution of the heavy metals in both the settled sludge and effluent was examined.

The digesters employed in this project and all subsequent studies were complete-mix, 10-litre volume units, such as shown in Figure 1. Similarly, the leachate feed for all aerobic studies closely resembled the composition shown in Table 6. All biological reactors were operated on a "fill and draw" basis, from start to steady state conditions. This biotreatment study was performed at room temperature (21° - 25°C) and at conventional BOD_5:N:P ratios of approximately 100:5:1.

Using high MLVSS concentrations, 8,000 to 16,000 mg/L, with a combination of air and mechanical mixing to control foaming, stable digester operation was maintained with sludge ages ranging from 10 to 60 days. Table 7 summarizes the overall performance of the digesters. For influent COD concentrations averaging 48,000 mg/L, settled effluent COD removal ranged from 96.0 to 99.2%, over a sludge age range of 10 to 60 days. Mixed liquor COD removals were correspondingly 51 to 76 percent. Similar treatment efficiencies were obtained for the BOD_5 results. Just increasing the sludge age from 10 to 20 days showed significant improvement in effluent quality. Settling was complete after 2 hours.

The leachate feed also contained a variety of metals in varying concentrations - iron exhibited the highest influent concentration at close to 1000 ppm. Most of these metals were almost completely removed by the settling biofloc; others were associated with the sludge solids to a lesser extent. Overall metal removals were as follows: aluminum, cadmium, calcium, chromium, iron, manganese and zinc - >95%; lead - average of 85%; nickel - average of 77%; magnesium - average of 60%; and potassium (with sodium) - average of 12%.

In this study, it was also observed that for the most part there existed a very predictable pattern of decreasing metal concentrations in the final effluent with increasing sludge age. At the same time, a drop in steady-state, mixed liquor metal concentrations was observed with increasing sludge age, despite the fact that corresponding MLVSS levels were also dropping. The indication is that sludge age or loading rate may play a more significant role than reactor solids levels in controlling metal concentrations. It is also recognized that since most of the metals were concentrated in the sludge, an additional problem has been created, with respect to further treatment and disposal of this sludge.

Analysis of the kinetic parameters in this study indicated that there was inhibition of the biological processes responsible for the removal of oxygen demanding material in this leachate - probably due to a combination of high metal concentration and exotic organic compounds. A recommendation for "best treatment" of this leachate (or any similar waste) called for a minimum sludge age of 20 days and a food to micro-organism ratio under 0.15 kg BOD_5/kg MLVSS/day.

(v) Project Title: Temperature Effects on Two-Stage, Aerobic Bio-Treatment of Leachate

Summary

This research project has just recently been terminated. To date, only preliminary data analysis has been completed - final results will be available, in report form, in early 1979 (13).

The aims of this study were threefold:

(a) two-stage biological treatment of a high-strength leachate with the second reactor acting as a polishing step,

(b) determination of overall effects on both reactors of a gradual temperature drop, and

(c) attempt to optimize the two-stage system under the varying conditions of temperature and sludge age.

Again, in this study, 10-litre volume reactors were employed and operated on a fill-and-draw basis under steady-state conditions. The mixed liquor was thoroughly mixed and aerated. Nutrient loading was slightly in excess of BOD_5:N:P=100:5:1. Sludge ages of 6,9, and 20 days were used in the first stage reactors, while the polishing reactors were operated at 6,10, 12 and 20 days. The reactors were started up and operated at room temperature (23-25°C) and then subsequently exposed to operating temperatures of 16°C and 9°C. The leachate was lysimeter-generated and was characterized by COD and BOD_5 values of 19,000 mg/L and 14,000 mg/L respectively; in addition, it contained a spectrum of metals (eg. iron at 1225 mg/L) and total solids over 10,000 mg/L.

Preliminary data analysis has revealed some interesting, and in certain cases, unexpected results. The removal of organic material in the first stage reactors was exceptionally good, with better than 99 percent BOD_5 and 95 percent COD removals being achieved. These results held true for most combinations of sludge age and temperatures tested. As a result of low, residual concentrations of biodegradable organics in the first-stage effluent, the polishing reactors were viable only at 9°C, the lowest temperature investigated. At this temperature and sludge ages examined, the polishing reactors further removed an average 80 percent of the residual BOD_5 and 30-60 percent of the COD. Second stage, steady-state MLVSS levels ranged from 150-400 mg/L only, while the first-stage units stabilized at MLVSS values of 4000 to 6000 mg/L, depending on sludge age.

First-stage metal removal was 90 percent or better for most of the nine metals monitored. Nickel and magnesium, however, experienced only an average 50% reduction. The polishing stage further reduced residual levels of manganese, iron and zinc while additional removal of the other metals was insignificant. This latter observation might be partially explained by the initial low concentrations of these metals in the first-stage effluent.

For the temperature ranges studied, the performance of the reactors, both in terms of organic and metal removal, was not significantly effected as long as the temperature adjustment was gradual. Only at 9°C was there indication of reactor instability, especially at the lower sludge ages. It was noted, however, that the settling characteristics of the mixed-liquor were highly variable, with a combination of higher sludge age and higher temperature producing a better settling effluent. At the low sludge ages and operating temperatures, mixed-liquor settling was slow and consistently produced highly-turbid effluent (SS ∿ 400 mg/L). Microscopic examination of the mixed-liquor revealed a network of branching, filamentous growth and a near absence of rotifers. In addition, some bacteria remained dispersed while others tended to form integrated, bacterial clumps.

(vi) Project Title: Operating Performance of Combined Biological-Chemical, Leachate Treatment Systems

Summary

This study was carried out in parallel to Project (v) and has also just been completed. The final report will be available in early 1979 (14).

This research project investigated the possibility of treating a high-strength, leachate with an aerobic, biological system followed by a lime-polishing step. Also investigated were the effects of varying temperature and sludge age, both separately and in combination, as well as lime dosages.

10-litre volume reactors were once again employed and operated on a daily fill-and-draw basis to steady-state conditions. Nutrient loadings were maintained at approximately BOD_5:N:P = 100:5:1. The leachate feed was similar to that employed in Project (v). Daily effluents being generated (after settling and/or filtering) were collected and properly stored for use in the polishing step. The investigation was carried out in two steps - a "temperature reduction phase, TRP" and a "cold-temperature phase, CTP". During the TRP portion, sludge ages of 25 and 15 days were employed in combination with temperatures of 24°C, 16°C and 9°C. In the CTP study, the operating temperature was lowered to 5°C and additional units of 12,9 and 6 day sludge age were adopted. This approach (in combination with the Project (v) work) thus provided a cross-section of operating possibilities for a single-stage biotreatment process, independent of the polishing step.

The lime-polishing step was performed under the Standard Jar Test procedure. To ensure uniformity, no lime testing was carried out until enough first stage effluent was collected to cover the entire second-stage testing schedule. Lime additions ranged from 100 mg/L to 1600 mg/L and were applied only to those effluents requiring further treatment (via prior analysis and examination of local pollution control objectives) (8).

Data analysis, to date, has revealed some interesting results. BOD_5 and COD removal efficiencies, during the TRP portion of this project, were exceptionally good - essentially better than 99% and 97% respectively, for both the filtered and settled effluents. Similarly, metal reductions were better than 90% for most of the metals monitored (eg. influent iron greater than 1000 mg/L). Lowering the operating temperature for the sludge ages investigated seemed to offer no particular operational problems - as a result, lime-polishing was determined as being unnecessary for many of the TRP effluents.

During the CTP portion of this work, reactor instability was observed, particularly at the lower sludge ages. Reactor failure was achieved at a combination of 5°C and 6-day sludge age. Lime-polishing was performed on both settled and filtered effluents from the remaining (and viable) 25,15,12, and 9 day units.

In most cases, the second stage or lime-polishing step consistently produced final effluents that were more than acceptable in meeting local discharge guidelines - in some cases even as low as 100 mg/L of lime additions. Further reduction in levels of organics, solids and metals were significant. However, residual organic and metal removals were found to be highly dependent on final pH and initial solids levels, with both influencing the level of metal precipitation achieved and quality of settling.

Again, mixed liquor settling proved highly variable, with progressively poor settling as operating temperatures and sludge ages decreased. Increasing the allowable settling time seemed to offer no particular advantages in these circumstances.

(vii) Project Title: Nutrient Requirements for Biostabilization of a Landfill Leachate

Summary

This project was an immediate follow-up study to the work of Uloth (10). The research has been completed and a final report is in progress (15).

This study investigated the nitrogen and phosphorus requirements of aerobic microorganisms treating a high-strength leachate, both by itself and in combination with domestic sewage (primary effluent). Special emphasis was placed on the efficiency of metal removal as a function of nutrient loading.

In the first series of experiments, the nutrient loading was varied in a set of biological reactors having a 20-day sludge age, operating at room temperatures (22-24°C), and fed with a high-strength leachate (as described previously). The BOD_5:N:P ratio was varied from 100:3.19:0.12 to 100:5:1.1. The most effective overall treatment was achieved with a loading of 100:3.19:1.11, as shown in Table 8. When the nutrient loading was increased or decreased from this level, there was a consistent increase in BOD_5, suspended solids and metal concentrations in the treated effluent. In addition, at nutrient loadings below 100:3.19:1.11, mixed liquor sludge bulking was prevalent.

In the second series of experiments, the leachate feed was combined with domestic sewage, in proportions varying from 0 to 20% (vol. leachate)/(vol. leachate plus sewage). No additional nutrients were added to the reactors - the BOD_5:N:P ratio varied from 100:24:4.3 to 100:3.62:0.12. The end result of this study was very effective treatment of all combined wastewaters. There was no corresponding increase in BOD_5 or metal concentrations in the treated effluent when a higher proportion of leachate was added to the reactors. The sludge did not bulk, even though the ratio dropped to 100:3.62:0.12.

As a result of this study, it appears that, if treating a similar leachate (or other waste) with a 20-day reactor or aerated lagoon, the required nutrient loading could be significantly reduced below the recommended BOD_5:N:P ratio of 100:5:1. At low volumetric BOD loadings of approximately 10 lb BOD_5 per day per 1000 cu ft, a low nutrient loading of 100:3.62:0.12 did not seem to adversely affect treatment efficiency. However, at a higher loading of 60 lb BOD_5 per day per 1000 cu ft, the minimum nutrient loading needed was 100:3.19:0.5. Therefore, an increase in volumetric loading of high-strength leachate required a corresponding increase in phosphorus input.

In addition, for the units that exhibited sludge bulking and poor settling under low nutrient loading conditions, a high level of suspended solids prevailed in the final effluent; because most of the metals were bound to these solids, final effluent metal concentrations were also dramatically increased.

(viii) Project Title: Preliminary Design, Richmond, B.C. Landfill

Summary

(This project is under the direction of R.D. Cameron).

A local landfill, receiving about 500 tons of municipal and commercial refuse per day, has been discharging leachate to a nearby river at an estimated rate of one million Imp gal per day. Leachate produced from the 310 acre active site is a combination of essentially all of the annual precipitation (40 in per year) plus river water entering at high tide. Having recently been brought under permit, the landfill operators are faced with having to treat the existing leachate, as well as that which will be produced during and after filling the remaining 350 acres.

Preliminary data (16) showed that the approximate reductions shown in Table 9 were required to meet effluent standards. With the expected site expansion, it was concluded that aerobic biological treatment, followed by activated carbon adsorption, and then followed by either reverse osmosis or ion exchange, would be the most feasible and flexible treatment method.

It was also expected that, with a proper collection system, less leachate dilution will be occurring with time; thus, leachate strength will increase. For this reason a leachate spring near the active face was chosen as the collection site for a relatively high-strength leachate. Table 10 shows the parameters exceeding effluent standards.

Semi-batch, aerated lagoon laboratory studies were conducted at 2, 3, 5, 7 and 10 day solids detention times. Nutrients were added to achieve a BOD:N:P ratio of 100:5:1. The results from this work are summarized in Table 11 (16). The higher strength second leachate sample also responded well to treatment with a BOD removal efficiency of 99.4% at 10 day solids detention time.

With the exception of iron, sulphate and boron, all contaminants were reduced to less than effluent standards. The effluent from treatment of leachate sample I was non-toxic, while that from sample II had a 96 h - LC_{50} of 72% by volume.

Insufficient leachate treatment effluent volume was available for complete activated carbon treatment. The few tests conducted, however, showed that iron could be reduced to less than effluent standards, using about 2½ to 3 times the carbon normally required for tertiary treatment. Overall COD removal efficiency was increased to 99.4%.

Small-scale, sludge-leaching tests were also performed to evaluate the possibility of placing the aerobically produced sludge back into the landfill. For just the metals, the percentage leached at a leaching volume of 212 mL per gm of dry sludge solids were:

Zn	< 0% (adsorbed from tap water)
Mn	0.4%
Fe	4.2%
Al	4.8%
Cr	14.5%

Subsequent to the research, the landfill operators called for a preliminary design. With the excellent BOD removal and good sludge settleability found in the laboratory study, enough confidence had been generated to recommend going to a full-scale facility without an intermediate pilot plant study. However, the rate of leachate production is expected to be such that the initial stages of the full scale facility will serve as a pilot study for subsequent stages if necessary. Thus, some reduction in leachate volume will probably occur initially, due to proposed changes in the leachate collection system. Within six or seven years, leachate strength and flow will peak at an expected BOD_5 of 5000 mg/L and a 1.3 Imp mgd flow rate.

Preliminary design will provide for a five day sludge age at a flow of 1 Imp mgd. While BOD removal efficiency will be somewhat reduced due to cooler field temperature, this will be balanced by both a flow reduction as well as a longer than necessary detention time. The aeration basin will be of the complete mix type with provision for additional units, if necessary; also provided will be separate clarifiers.

It is also expected that "tees" will be provided from the sludge lines, so that sludge recycle can be practised. Sufficient space will be allotted for polymer or chemical addition, before settling, although this is not considered to be likely. Space will also be provided for activated carbon treatment (or something else) of the effluent.

Preliminary agreement has been reached with the regulatory agency such that, if all other parameters meet the effluent guidelines, the sulphate, iron and boron standards will be waived. All sludge will be returned to the landfill for disposal without dewatering.

This flexible design, if installed, will provide valuable operating data which can then be used to evaluate the field kinetic coefficients. If the optimistic design criteria are not suitable for the expected increased loading, sufficient lead time will be available to add to the initial facility. The facility will also be designed so that, for the aeration basins at least, stages can be removed as the site ages and the space utilized for the ultimate site development.

Two major questions are currently unanswered - one will be answered during field operation while the other will not. The first question is whether or not metal leaching from the returned sludge will be sufficient to cause a plant overload. The second question relates to where the sludge will be disposed of after the landfill operation has ceased. This will likely be a problem for ten to fifteen years and is one which has not yet been addressed.

SUMMARY AND CONCLUSIONS

As discussed in the preceeding pages, a broad spectrum of possible leachate treatment schemes has been examined and the resulting data fully or partially analyzed. Certainly many of the questions and uncertainties surrounding leachate treatment at the start of this long-term project have been eliminated and the project scope narrowed down considerably.

However, there are still some treatment aspects requiring further examination as a result of those projects completed to date. Further bench-scale research is therefore planned,especially in the areas of "treatment sludge", combined treatment and effluent polishing. Although these areas do need further investigation, preliminary optimization and scale-up is not too unrealistic at this time, depending on the characteristics of the leachate and circumstances in question. To this end, further work is now in progress.

ACKNOWLEDGEMENTS

The author is indebted to his colleague, Dr. R.D. Cameron - it is through much of Bob's efforts that this project was initiated and is continuing. In addition, the author is grateful to the technical staff, especially Mrs. E. McDonald, head technician, and many graduate students that have been involved with this study over the years. Financial support for this continuing project has originated from the Government of British Columbia, National Research Council of Canada, Environment Canada, and the City of Vancouver.

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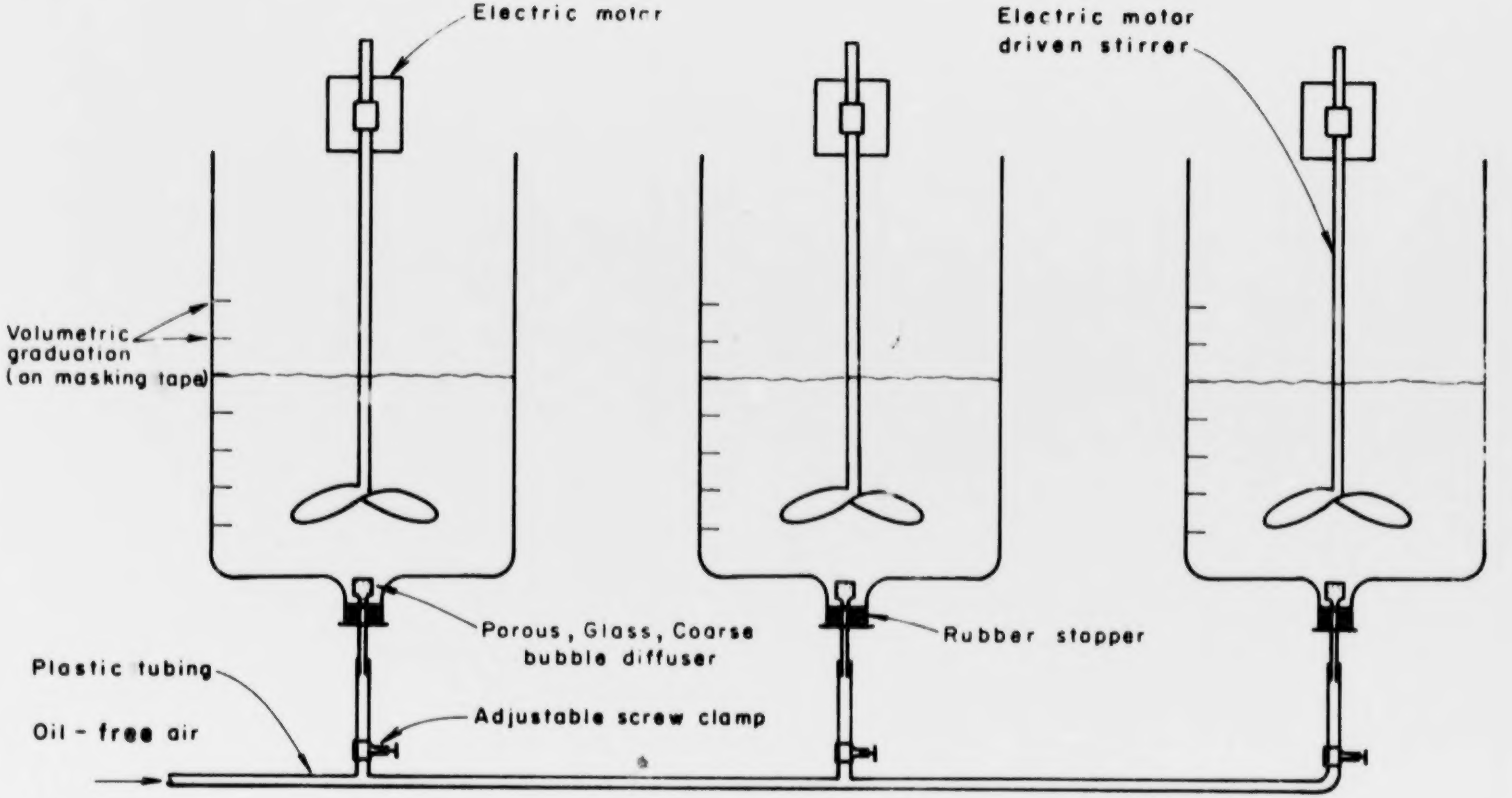
Electric motor
Electric motor driven stirrer
Volumetric graduation (on masking tape)
Porous, Glass, Coarse bubble diffuser
Rubber stopper
Plastic tubing
Oil - free air
Adjustable screw clamp

FIGURE 1 : SCHEMATIC OF LABORATORY AEROBIC DIGESTERS.
(10 litre volume units)

TABLE 1

COMPOSITION OF TYPICAL LEACHATES

Parameter	Range of Values or Concentrations* (Landfills and Test Lysimeters)
BOD_5	9 - 55,000
COD	0 - 90,000
Total Carbon	715 - 22,350
Total Organic Carbon	715 - 22,350
Total Solids	1,000 - 45,000
Total Volatile Solids	1,000 - 23,157
Total Dissolved Solids	0 - 42,300
Acidity	0 - 9,560
Alkalinity	0 - 20,900
Aluminum	0 - 122
Arsenic	0 - 11.6
Barium	0 - 5.4
Beryllium	0 - 0.3
Calcium	5 - 4,000
Cadmium	0 - 0.19
Chloride	34 - 2,800
Chromium	0 - 33.4
Copper	0 - 10
Iron	0.2 - 5,500
Lead	0 - 5.0
Magnesium	165 - 15,600
Manganese	0.06 - 1,400
Mercury	0 - 0.064
Molybdenum	0 - 0.52
Nitrogen - total	0 - 2,406
- NH_3	0 - 1,106
Nickel	0.01 - 0.80
Phosphorus - total	0 - 154
Potassium	2.8 - 3,770
Sodium	0 - 7,700
Sulphates	1 - 1,826
Sulphides	0 - 0.13
Titanium	0 - 5.0
Vanadium	0 - 1.4
Zinc	0 - 1,000
pH	3.7 - 8.5
Tannin-like compounds	78 - 1,278
Colour (chloroplatinate)	0 - 12,000
Odour	not detectable to terrible

*all values except those for pH, colour and odour are in mg/L.

TABLE 2

TYPICAL LYSIMETER DETAILS

(one of 16 such units)

(1) Dimensions - 14 feet (4.27 m) deep, 4 feet (1.22 m) in diameter

(2) Cover Material - 2 feet (0.61 m) of hog fuel

(3) Total weight of refuse - 3420 lb (1549 kg)

(4) Depth of refuse - 8 feet (2.44 m)

(5) Weight (density) before final cover - 884 lb/cu yd (wet) (523 kg/m^3)

(6) Simulated Precipitation - 15 inches per year (38.1 cm/yr)

(7) Moisture content - 34.7%

(8) Percentage composition of refuse:

Food waste - 11.8
Garden waste - 9.8
Paper products - 47.6
Cardboard - 5.4
Textiles - 3.6
Wood - 4.7
Metals - 8.7
Glass and ceramics - 7.0
Ash, rocks and dirt - 1.4

Total - 100%

TABLE 3

A TYPICAL LEACHATE ANALYSIS FOR PROJECT (i)

pH	-	5.3
Colour	-	2500 Helige colour units
Total Solids	-	7140 mg/L
Turbidity	-	69 Hach Formazin Units
Total Carbon	-	5230 mg/L
Organic Carbon	-	5203 mg/L
COD	-	14,000 mg/L
Fe	-	47 mg/L
Zn	-	12.45 mg/L
Na	-	188 mg/L
K	-	156 mg/L
Ni	-	0.165 mg/L
Cu	-	0.39 mg/L
Mn	-	10.14 mg/L
Cr	-	1.14 mg/L
Al	-	0.40 mg/L
Si	-	36.3 mg/L
Co	-	2.10 mg/L

TABLE 4

LEACHATE ANALYSIS - IN SITU SAMPLE FOR PROJECT (ii)

Parameter	Measurement
pH	7.1
Alkalinity (pH = 4.5)	3400
COD	903
Total residue	4636
Non-filtrable residue	134
Calcium	175
Chloride	2400
Iron	30.3
Lead	0.065
Magnesium	126
Manganese	0.57
Nitrogen - total Kjeldahl	494
Phosphorus	1.56
Potassium	600
Sodium	840
Zinc	0.43
BOD_5	120
Acidity (pH = 8.3)	185
Volatile solids	1092
Fluoride	0.27
Nitrogen - ammonia	427
Sulphate	5.3
Tannin-like compounds	62.4
Aluminum	0.27
Arsenic	0.038
Barium	0.08
Beryllium	0.025
Boron	4.5
Cadmium	0.0037
Chromium	0.053
Cobalt	N.D.
Copper	0.024
Molybdenum	0.013
Nickel	0.069
Silicon	9.80
Silver	N.D.
Tin	N.D.
Titanium	N.D.
Vanadium	N.D.

Notes: N.D. - not detectable. All values except pH as mg/L; alkalinity and acidity in mg/L as $CaCO_3$. All metals as total concentrations.

TABLE 5

PEAT REQUIREMENTS FOR LEACHATE TREATMENT - PROJECT (ii)

	pH		
	7.1	7.8	8.4
Total dissolved solids loading (mg/g)	34.1	16.2	35.3
Total metal removal (mg/g)	12.6	5.0	13.1
Peat required (kg(dry)/1000L leachate)	159	397	159
COD removal (mg/g)	5.7	2.4	3.9
P removal (mg/g)	0.01	0.005	0.07
TKN removal (mg/g)	3.5	1.6	3.0
Non-filtrable residue removal (mg/g)	0.9	0.4	2.1
Chloride removal (mg/g)	16.3	6.4	15.6

TABLE 6

LEACHATE FEED COMPOSITION FOR PROJECT (iv)

BOD_5	- 36,000 mg/L	Cadmium	- 0.39 mg/L
COD	- 48,000	Chloride	- 1,620
Total Carbon	- 15,400	Chromium	- 1.9
Total Org. Carbon	- 15,389	Copper	- 0.24
Total Solids	- 26,600	Iron	- 960
Tot. Volatile Solids	- 17,800	Lead	- 1.44
Tot. Dissolv. Solids	- 25,700	Magnesium	- 310
Acidity	- 5,640	Manganese	- 41.0
Alkalinity	- 7,640	Mercury	- trace
Aluminum	- 41.8	Nitrogen - Total	- 1,080
Arsenic	- 3.6	- NH_3	- 725
Barium	- 0.7	Nickel	- 0.65
Beryllium	- trace	Phosphorus - total	- 19.8
Boron	- 7.30	Potassium	- 1,060
Calcium	- 1,394	Sodium	- 1,250
		Selenium	- 0.450
Tannin-like Cmpds.	- 578	Sulphates	- 1,070
pH	- 5.02	Zinc	- 223

TABLE 7

TREATMENT EFFICIENCY AND REACTOR CHARACTERISTICS FOR PROJECT (iv)

Parameter (all but pH in ppm)	Digester Sludge Age, θ_c (days)					Leachate Feed
	10	20	30	45	60	
Steady-State MLSS Concentration	24,250	22,650	20,175	20,300	14,300	-
Steady-State MLVSS Concentration	16,100	15,100	12,045	11,880	8,100	-
Effluent Total Solids	6,050	5,200	5,070	4,870	4,450	26,600
Effluent COD	1,547	594	533	428	386	48,000
Effluent BOD_5	129	32	59	66	75	36,000
Effluent Total Organic Carbon	683	268	221	-	-	15,389
Effluent Alkalinity	1,320	1,210	969	728	542	7,640
Effluent pH	8.80	8.73	8.65	8.74	8.60	5.02

TABLE 8

PARTIAL DATA SUMMARY FOR LEACHATE TREATMENT STUDY - PROJECT (vii)

	Leachate Feed	Reactor A	Reactor B	Reactor C	Reactor D	Reactor E	Reactor F
(1) Treated Effluent (mg/L)							
BOD_5	19330	82	55	36	300	1430	560
S.S.	990	380	133	47	1305	245	160
Chromium	0.365	0.050	0.033	0.035	0.103	0.040	0.033
Iron	960	25.2	9.72	4.27	27.3	13.5	6.73
Lead	0.167	0.011	0.006	0.003	0.023	0.005	0.015
Zinc	49.5	1.31	0.630	0.295	2.10	0.726	0.543
(2) Nutrient Loading							
BOD/N/P		100 5.03 1.11	100 3.98 1.11	100 3.19 1.11	100 3.98 0.32	100 3.98 0.12	100 3.19 0.12

TABLE 9

LEACHATE STRENGTH REDUCTION REQUIRED FOR PROJECT (viii)

Parameter	Effluent Standard	% Reduction Needed
BOD_5	100 mg/L	87
SS	100	50
Sulphate	50	75
Iron	0.30	99.4
Lead	0.05	50
Manganese	0.05	99
Zinc	0.50	60
Toxicity (96 hr LC_{50})	100%	80

TABLE 10

LEACHATE COLLECTED FOR PROJECT (viii)

Parameter	Sample 1	Sample 2
BOD_5	1140 mg/L	2980 mg/L
SS	125	NA
Sulphate	250	83
Boron	5.9	7.4
Iron	22.4	1.6
Manganese	4.3	7.8
Zinc	1.3	0.6
96 hr LC_{50}	NA	4.2% V/V

TABLE 11

AEROBIC LAGOON TREATMENT OF LEACHATE - PROJECT (viii)

Influent BOD_5 - 1000 mg/L (approx.)
Growth yield coeff. - y - 0.59 mg VSS/mg BOD_5
Endogenous decay coeff. - b - 0.040 day^{-1}
k_s - 99 mg/L
k - 4.5 day^{-1}
θ_c min. - 0.42 days
BOD_5 removal - 99.1%
COD removal - 88%
MLVSS - 560 mg/L @ θ_c = 5 days
F/M ratio - 0.4 day^{-1} @ θ_c = 5 days
Settled effluent SS - 10 - 20 mg/L

LEACHATE TREATMENT DEMONSTRATION

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ABSTRACT

This paper reports the results of three years of operation of a solid waste landfill leachate treatment plant. This plant incorporates chemical, physical, and biological unit processes and was constructed to enable operation in several sequences of unit processes. The processes include: equalization, lime precipitation, sedimentation, ammonia stripping, activated sludge, carbon adsorption, and chlorination.

The purpose of this project was to demonstrate the treatment of landfill leachate at full scale in four treatment sequences. The plant is supplied by an adjacent, lined landfill which generates about 20,000 gallons per day of leachate high in organic matter, ammonia, and heavy metals.

Of the treatment sequences demonstrated, that sequence which incorporates equalization, lime precipitation/clarification, ammonia stripping, activated sludge treatment, and chlorination (in that order) has been the most effective in achieving the discharge standards for this wastewater treatment facility. This treatment sequence has been found capable of meeting the discharge standards except BOD and lead, which have been occasionally exceeded, and ammonia, which has seldom been achieved.

BACKGROUND

EPA has been studying the treatment of landfill leachate, because of the potential for adverse effect on ground and surface waters, for several years. Most investigations were laboratory-scale or occasionally pilot-scale treatment systems. In early 1975 the Office of Solid Waste became aware of initiation of construction of a leachate treatment system to serve a lined landfill in Pennsylvania. That State had awarded an operating permit for a mixed municipal solid waste landfill which was to be constructed above grade, underlain with and surrounded by a dike covered with a sprayed asphaltic compound, in 1970. The permit included a provision requiring leachate treatment and discharge to the Delaware River at such time as the dike became filled with leachate to within two feet of overflowing. The Delaware River Basin Commission subsequently (1974) awarded a permit for discharge of treated leachate from the landfill to the Delaware River during the high stream flow months of December through April in accordance with the effluent quality criteria listed in Table 1

Applied Technology Associates was awarded a demonstration grant in late 1975 to evaluate the efficiency and economics of full-scale treatment of landfill leachate at the GROWS (Geological Reclamation Operations and Waste Systems, Inc.) landfill, located in Bucks County near Tullytown, Pennsylvania. The evaluation was to comprise two years of full-scale

TABLE 1
SUMMARY OF EFFLUENT CRITERIA*

Parameter	Maximum Concentration mg/liter
BOD_5	100.0
Ammonia-Nitrogen	35.0
Phosphate	20.0
Oil and grease	10.0
Iron	7.0
Zinc	0.6
Copper	0.2
Cadmium	0.02
Lead	0.1
Mercury	0.01
Chromium	0.1

*Commonwealth of Pennsylvania Department of Environmental Resources and Delaware River Basin Commission.

operation of this leachate treatment system which incorporated chemical, physical and biological treatment.

THE SYSTEM

The leachate treatment system designed to meet the needs of the GROWS landfill was a modification of a package type sewage treatment plant. The system consisted of a flash mixing chamber for lime addition to influent leachate prior to introduction to a cylindrical clarification unit. Clarified effluent was then to be introduced to a prefabricated aeration/sedimentation unit for activated sludge biological treatment, prior to discharge.

In order to determine design parameters for the leachate treatment system, samples of the landfill leachate were obtained and used for operation of a laboratory bench-scale treatment system. The leachate characteristics upon which plant design was based are presented in Table 2. Limited sampling in the early stages of landfill operation yielded these characteristics which will be shown later as having changed dramatically. The ultimate design flow for this system was estimated by a water balance at 100 gpm.

During the three years of this demonstration the original design has been modified significantly through addition of two lagoons. The first lagoon was constructed to serve as an ammonia stripping pond following lime addition to the influent leachate. The second lagoon, added late in the project, serves as an equalization pond constructed ahead of the entire leachate treatment system.

The current facility is shown schematically for System 1 in Figure 1. Detention times in the various units are about 2 days in each of the lagoons, 2 hours in the reactor clarifier, and 6 hours in the aeration chambers at the design flow of 100 gpm. Since current flow rates are only about 15 gpm these detention times should be increased accordingly.

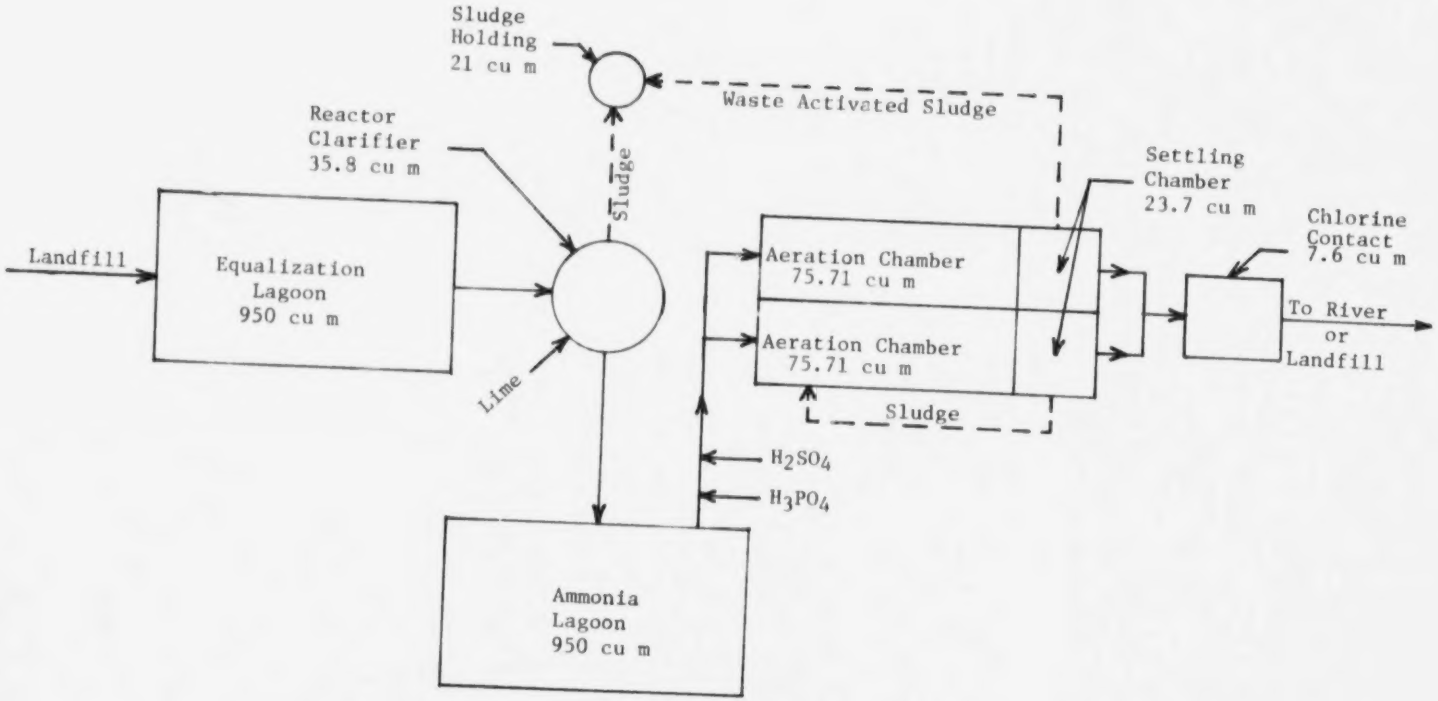
As a result of award of the demonstration grant the facility has been equipped with a laboratory trailer to enable prompt analysis of samples to enable system evaluation and operation.

TABLE 2
DESIGN LEACHATE CHARACTERISTICS

Constituents	Raw Leachate*
BOD_5	1500
Sus. Solids	1500
Total Solids	3000
Percent Volatile	55
pH	5.5
Chlorine	200
Iron, total	600
Zinc	10
Chloride	800
Organic Nitrogen	100
Nitrate	20
Sulfate	300
Copper	1
Hardness	800
Alkalinity	1100
Color, standard units	50
Flow, mgd	0.144
Temperature, °F	80

*All units are mg/l except pH, color, flow, and temperature.

FIGURE 1. SCHEMATIC FLOW SYSTEM 1


Sludge
Holding
21 cu m
Waste Activated Sludge
Reactor
Clarifier
35.8 cu m
Sludge
Settling
Chamber
23.7 cu m
Chlorine
Contact
7.6 cu m
Landfill
Equalization
Lagoon
950 cu m
Aeration Chamber
75.71 cu m
Aeration Chamber
75.71 cu m
To River
or
Landfill
Lime
Sludge
H_2SO_4
H_3PO_4
Ammonia
Lagoon
950 cu m

DISCUSSION

As previously indicated the design leachate characteristics presented in Table 2 have been shown to be highly inaccurate. Over the life of this project leachate volume and strength have fluctuated widely but have tended to increase overall. Table 3 presents the quality of the raw landfill leachate for each year of this project and the average leachate quality, over time. To further depict the variability of raw leachate quality Figure 2 is included. As seen in Figure 2, the COD has varied from as low as 5000 mg/l to as high as 50,000 mg/l.

Anyone who has worked with landfill leachate is familiar with such fluctuations in leachate quality. Elaboration on this topic is considered appropriate in this report in that it demonstrates one of the problems encountered by both the designer and operator of such a leachate treatment facility. Since, as in this case, it is often necessary to design the leachate treatment system either at the design stage of the landfill or in the very early stages of landfill operation, the designers must rely upon the literature for "typical" leachate quality modified by leachate quality prediction techniques. In this situation, young landfill leachate and prediction techniques were inadequate.

From the operator's standpoint leachate quality variations can cause serious difficulty in attempting to automate chemical addition to the influent raw leachate, such as lime. Only frequent analysis of influent leachate and subsequent adjustment of chemical feed can accomplish the desired result. Significant variation in the organic strength of influent leachate to a biological treatment element is also a problem, but the buffering capacity of activated sludge ameliorates this problem somewhat. Construction of an equalization lagoon upstream of the treatment facility was the "solution" selected to reduce the degree of quality fluctuation at this facility.

Construction of this leachate treatment facility was completed in late 1975. Start-up attempts were initially frustrated by freezing temperatures until Spring of 1976. Three attempts at establishing an activated sludge culture by acclimation of waste activated sludge "seed" from a nearby sewage treatment plant were unsuccessful. Bench-scale simulation of the biological unit process indicated that the landfill leachate was deficient in phosphorous and had an excessive ammonia concentration which was thought to be toxic to the biomass. The phosphorous deficiency was immediately corrected by addition of phosphoric acid ahead of the aeration tanks. In order to deal with the high concentration of ammonia, the twin aeration tanks were modified from parallel to series flow. In the first tank the pH was raised to greater than 10 to enable ammonia stripping by aeration. After several weeks of operating in this mode, it was concluded that there was insufficient surface area for effective ammonia stripping. This decision led to construction of the first of the two "add-on" lagoons.

Construction of this lagoon equipped with a diffused aeration system proved effective in reducing influent ammonia concentrations by greater than 50%. A leachate acclimated biomass was finally achieved in July 1976. This biomass was again lost to freezing weather during the 1977-78 winter and reestablishment of the biomass in the Spring of 1978 was somewhat less difficult.

As indicated earlier, chemical/physical followed by activated sludge treatment was the original treatment design and the sequence expected to be the most effective. This sequence (referred to as System 1) was initiated on August 1, 1976 (following acclimation of the activated sludge to leachate) and evaluated through April 30, 1977.

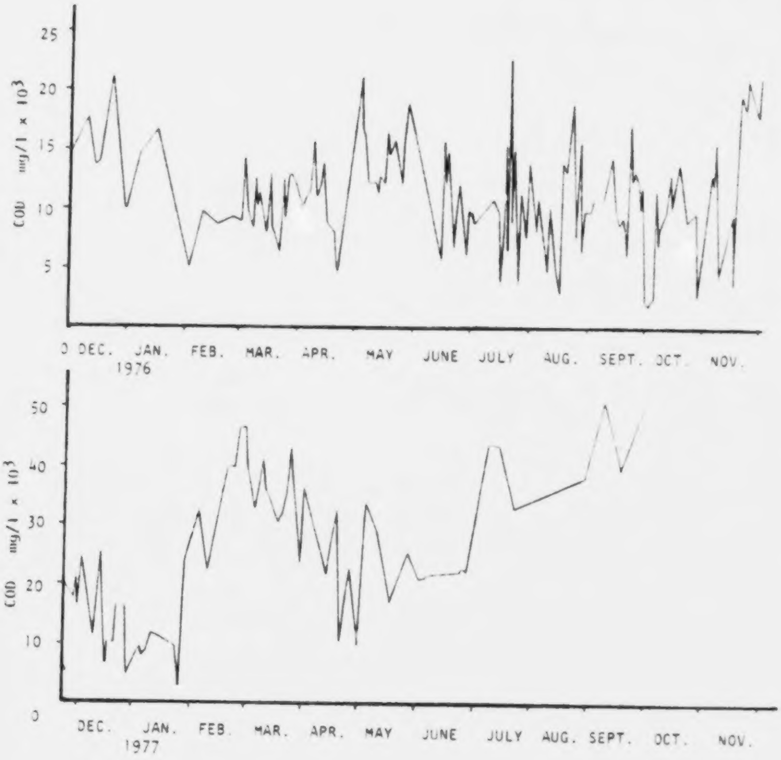
Operation in the System 1 sequence involves chemical addition, primarily lime, to the raw influent leachate to precipitate dissolved metals and reduce suspended solids. Following clarification the leachate, now at a pH in excess of 10.5, is introduced into the aerated ammonia stripping lagoon. The lagoon effluent is neutralized with acid, if necessary, and supplemented

TABLE 3. LANDFILL LEACHATE CHARACTERISTICS*

Item	11/15/75-9/1/76	Concentration+ 9/1/76-9/1/77	9/1/77-8/31/78	11/15/75-8/31/78
Biochemical oxygen demand (5-day)	4,460	13,000	11,359	10,907
Chemical oxygen demand	11,210	20,032	21,836	18,553
Suspended solids	1,994	549	1,730	1,044
Dissolved solids	11,190	14,154	13,181	13,029
pH	7.06	6.61	7.31	6.85
Alkalinity, as $CaCO_3$	5,685	5,620	4,830	5,404
Hardness, as $CaCO_3$	5,116	4,986	3,135	4,652
Calcium	651	894	725	818
Magnesium	652	454	250	453
Phosphate	2.81	2.61	2.98	2.74
Ammonia-N	1,966	724	883	1,001
Kjeldahl-N	1,660	760	611	984
Sulfate	114	683	428	462
Chloride	4,816	4,395	3,101	4,240
Sodium	1,177	1,386	1,457	1,354
Potassium	969	950	968	961
Cadmium	0.043	0.09	0.10	0.086
Chromium	0.158	0.43	0.22	0.28
Copper	0.441	0.39	0.32	0.39
Iron	245	378	176	312
Nickel	.531	1.98	1.27	1.55
Lead	.524	0.81	0.45	0.67
Zinc	8.70	31	11.0	21
Mercury	.0074	.0051	0.012	0.007

*These values represent the arithmetic mean of all raw leachate data.
+All units mg/liter except pH.

FIGURE 2. RAW LEACHATE CHEMICAL OXYGEN DEMAND
(Note change in scale on ordinate.)



with phosphoric acid prior to introduction into one of the two parallel activated sludge chambers. Clarified effluent from the biological system is either returned to the landfill or discharged to the Delaware River depending upon effluent quality and the time of year. Effluent is chlorinated before discharge to the Delaware River.

This same System 1 sequence was retested in July and August 1978 with the following modifications:

1. raw leachate was first introduced into an aerated equalization lagoon,

2. the activated sludge aeration chambers were operated in series instead of in parallel, and

3. the flow rate was only about half that of the previous operating period (i.e. 10,000 vs 22,000 gpd).

Concurrent evaluation of System 2 also took place during these periods since the System 2 sequence is chemical/physical treatment only. The performance of System 2 is discussed later in this report.

Table 4 displays the leachate treatment efficiency of the two operating periods in the System 1 sequence in comparison to the discharge standards. As seen in this table BOD removal exceeded 99% during the first operating period but only approached 95% during the second. Conversely, ammonia removal during the later period exceeded 99% while during the earlier period of operation only 89% ammonia removal was achieved, on the average. Finally, COD removal approached 95% during both these operating periods. No satisfactory explanation for these variations in performance has been developed.

Presented in Table 5 are the average costs of operation and maintenance of this treatment facility in the System 1 sequence. You will note in this table that power costs account for nearly two-thirds of the cost of treatment. This high power costs reflects the demand for leachate and effluent pumping and maintenance of the laboratory in addition to the requirements for actual treatment. In this treatment process, the aeration blowers account for the greater energy consumption. Labor of approximately 20 man-hours per week is not included in these costs. Please note that costs presented in this table are the averages of data collected only for these two periods of operation.

TABLE 5
SYSTEM 1 - OPERATION AND MAINTENANCE COSTS
(8/1/76 - 5/1/77; 7/1/78 - 8/31/78)

Total Flow, gal	5,953,255
Lime, lb/1000 gal	36.43
Sulfuric acid, gal/1000 gal	0.132
Phosphoric acid, gal/1000 gal	0.019
NaOH, gal/1000 gal	0.123
NaOCl, gal/1000 gal	0.151
Costs, $/1000 gal	
Power	1.92
Lime	1.10
H_2SO_4	0.10
H_3PO_4	0.05
NaOH	0.08
NaOCl	0.11
Total	3.36

Leachate treatment in the System 2 sequence was evaluated over a longer period of time than by any other sequence. Since System 1 includes System 2, information on this chemical/physical treatment sequence was gathered during the periods of operation in System 1. In addition, except during the brief periods of evaluation in the Systems 3 and 4 treatment sequences, evaluation of the treatability of raw leachate by System 2 was continued.

The System 2 treatment sequence has essentially been evaluated in three modes. Since System 2 data gathering began prior to accilimation of the activated sludge biomass, some data has been gathered on the effectiveness of lime addition to the raw leachate followed by clarification. The majority

TABLE 4. SYSTEM 1* TREATMENT PERFORMANCE AFTER ACCLIMATION OF ACTIVATED SLUDGE (August 1, 1976 - May 1, 1977; and July 1, 1978 - August 31, 1978)

	8/1/76 - 5/1/77			7/1/78 - 8/31/78			
	Concentration			Concentration			
Parameter	Influent**	Effluent**	Percent Removal	Influent**	Effluent**	Percent Removal	Discharge Standard
Sus. Solids	686	101	97.4	1655	478	71.1	
Dis. Solids	13563	5693	58.0	13091	7244	44.7	
COD	18488	939	94.9	18505	1008	94.6	
BOD_5	12468	118	99.1	8143	464	94.3	100
Alkalinity	5479	685	87.5	5262	1496	71.6	
Hardness	5331	1314	75.4	2504	1456	41.9	
Magnesium	499	107	78.6	275	105	62.0	
Calcium	929	347	62.6	653	113	82.7	
Chloride	4264	2592	39.2	8578	2254	73.7	
Sulfate	645	951	---	178	836	---	
Phosphate	2.15	13.7	---	1.39	17.2	---	
Ammonia-N	705	80	88.7	1076	6.3	99.4	35
Kjeldahl-N	748	102	86.4	---	---	---	
Sodium	1310	821	37.3	1248	1145	8.3	
Potassium	906	524	42.2	872	743	14.8	
Cadmium	0.08	0.01	87.5	0.06	0.04	33.3	0.02
Chromium	0.28	0.07	75.0	0.16	0.04	75.0	0.1
Copper	0.44	0.10	77.3	0.20	0.16	25.0	0.2
Iron	376	3.0	99.2	9.7	0.71	99.3	7.0
Nickel	1.91	0.76	60.2	0.88	0.67	24.1	
Lead	0.82	0.12	85.4	0.30	0.11	64.6	0.1
Zinc	22	0.57	97.4	3.38	0.16	95.3	0.6
Mercury	0.006	.004	28.9	0.003	0.002	33.3	0.01
Flow, gpd	21034			10000			

---

* This system consists of lime addition, sedimentation, air stripping, neutralization, nutrient supplementation and activated sludge.

** mg/l

of data on this treatment sequence was gathered when the system included chemical addition, primarily lime, to both assist in heavy metal removals and clarification, as well as, raise the pH to above 10.5 to enable ammonia removal by stripping in the aerated lagoon. Data gathering in the final mode followed installation of the aerated equalization lagoon at the head of the plant. Leachate, when treated only in the System 2 sequence, was returned to the landfill.

In Table 6 is presented the performance data on System 2. Values contained in this table are the averages of all samples analyzed. As indicated, influent values are the averages for the entire project period. Lime treatment effluent values are those collected whenever this process was in operation. Finally, ammonia stripping lagoon effluent values are those obtained whenever it was operating. For this reason effluent values should be compared only to influent and not to one another except in a general way.

The information in Table 6 generally indicates the beneficial effect of the ammonia stripping lagoon. As seen, ammonia was reduced by nearly 60 percent when the lagoon was in operation.

Presented in Table 7 are the average costs of operation and maintenance of this treatment facility in the System 2 sequence. As seen in this table, cost information is presented for three operating periods, one without and two with the ammonia stripping lagoon in operation. As with System 1 power costs represent the major cost of leachate treatment. Because of fluctuations in the raw leachate quality and the fact that the operating periods do not overlap, the two total cost figures should not be directly compared.

Following completion of the initial evaluation of System 1 on May 1, 1977, a four-month evaluation of treatment sequences, Systems 3 and 4, was conducted. System 3 consisted of reversing the flow so that raw leachate first entered the biological treatment tanks followed by chemical/physical polishing of the effluent. System 4 represents the treatment efficiencies of the biological treatment section alone. These two treatment sequences were attempted only after the activated sludge had become acclimated to leachate, since early project attempts to acclimate a sewage activated sludge to raw leachate had failed. Table 8 summarizes the efficiencies of these two sequences in treating landfill leachate.

TABLE 7. OPERATION AND MAINTENANCE COSTS - SYSTEM 2 - (11/15/75 - 5/1/77 and 11/1/77 - 8/31/78)

	During Operation Without Lagoon 11/15/75 - 6/14/76	During Operation With Lagoon 6/14/76 - 5/1/77 & 11/1/77 - 8/31/78
Flow, average gpd	22,805	38,618
Lime, lb/1000 gal	29.7	19.40
NaOH, gal/1000 gal	0	0.044
NaOCl, gal/1000 gal	0	0.054
Costs, $/1000 gal		
Power	1.48	1.70
Lime	.89	0.58
NaOH	0	.03
NaOCl	0	.04
Total	2.37	2.35

TABLE 6. SUMMARY OF EFFECTS OF CHEMICAL/PHYSICAL TREATMENT

	Raw Leachate	Lime Treatment Effluent +		Ammonia Lagoon Effluent ++	
	$\bar{x}$(mg/l)	$\bar{x}$(mg/l)	R(%)	$\bar{x}$(mg/l)	R(%)
Sus. Solids	1044	239	77.1	288	72.4
Dis. Solids	13029	7972	38.8	4650	64.3
COD	18553	7188	61.2	8793	52.6
BOD_5	10907	5265	51.7	3600	67.0
Alkalinity	5404	3052	43.5	2374	56.1
Hardness	4652	2461	47.1	1587	65.9
Magnesium	453	209	53.9	117	74.2
Calcium	818	696	14.9	424	48.2
Chloride	4240	3516	17.1	2669	37.0
Sulfate	462	426	7.8	525	--
Phosphate	2.74	0.26	90.5	0.27	90.1
Ammonia-N	1001	890	11.1	412	58.8
Kjeldahl-N	984	867	11.9	349	64.5
Sodium	1354	830	38.7	956	29.4
Potassium	961	613	36.2	572	40.5
Cadmium	0.086	0.03	65.1	0.04	53.5
Chromium	0.28	0.09	67.8	0.08	71.4
Copper	0.39	0.10	74.7	0.27	30.8
Iron	312	3.8	98.8	5.6	98.2
Nickel	1.55	0.57	63.2	0.73	52.9
Lead	0.67	0.24	64.2	0.23	65.7
Zinc	21	0.61	97.1	0.85	96.0
Mercury	0.007	0.003	57.1	0.010	--

+ 11/15/75 - 5/1/77 and 11/1/77 - 8/31/78.

++ 6/14/76 - 5/1/77 and 11/1/77 - 8/31/78.

$\bar{x}$ Average concentration.

R Percentage removal.

As seen in this table biological treatment only, while showing significant reductions in organic matter, did not perform anywhere well enough to permit discharge in compliance with the facilities discharge permit. System 3, which included chemical/physical polishing of the effluent, achieved significantly greater removal efficiencies but again failed to reach discharge standards. No cost information was developed for these treatment sequences.

The overall conclusion, based upon system performance in the four described sequences, is that treatment of leachate from this landfill must include biological treatment preceded by chemical/physical processes.

Of the several additional leachate treatment technologies investigated in bench-scale units on a limited basis, only activated carbon filtration and breakpoint chlorination yielded meaningful results. Carbon filtration of this raw leachate was unsuccessful due to rapid plugging of the filter media. Carbon filtration of system effluent, however, provided significant additional reductions in heavy metals concentrations. Breakpoint chlorination was demonstrated as accomplishing additional ammonia reduction in the final treatment plant effluent. The remaining limited evaluations of methods for improved heavy metals and suspended solids removal were inconclusive.

A more complete discussion of these bench-scale evaluations, as well as a thorough discussion of performance in the Systems 1 through 4 treatment sequences, is included in the final report on this demonstration. This report, entitled "Demonstrating Leachate Treatment - Report on a Full-scale Operating Plant," will be available shortly from the Office of Solid Waste.

TABLE 8. SUMMARY OF SYSTEMS 3 AND 4 (5/1/77 - 8/31/77)

Parameter	Raw Leachate $\overline{x}$ (mg/l)	System 4 Effluent $\overline{x}$ (mg/l)	System 4 Effluent R(%)	System 3 Effluent $\overline{x}$ (mg/l)	System 3 Effluent R(%)	Discharge Standard (mg/l)
Alkalinity	5087	2788	45.2	1178	76.8	*
Ammonia-N	649	312	51.9	153	76.4	35
BOD$_5$	12649	2150	83.0	763	94.0	100
Cadmium	0.11	0.08	27.3	0.02	81.8	0.02
Calcium	937	573	38.8	287	69.4	*
Chloride	4178	3778	9.6	1496	64.2	*
Chromium	0.48	0.37	22.9	0.08	83.3	0.1
COD	21152	4680	77.9	2257	89.3	*
Copper	0.27	0.22	18.5	.07	74.1	0.2
Dis. Solids	14742	10081	31.6	5353	63.7	*
Hardness	4463	2805	37.1	924	79.3	*
Iron	348	195	44.0	1.02	99.7	7
Kjeldahl-N	708	347	51.0	180	74.6	*
Lead	0.76	0.50	34.2	0.15	80.3	0.1
Magnesium	350	242	30.9	48	86.3	*
Mercury	0.007	.007	0	.002	71.4	0.01
Nickel	2.0	1.29	35.5	0.27	86.5	*
Phosphates	2.3	4.6	--	0.56	75.7	*
Potassium	1076	996	7.4	476	55.8	*
Sodium	1536	1412	8.1	719	53.2	*
Sulfate	658	853	--	513	22.0	*
Sus. Solids	1136	1322	--	180	84.2	*
Zinc	40	19	52.5	0.51	98.7	0.6

* No discharge standard for this parameter.
$\overline{x}$ Average concentration. R Percentage removal.

REMEDIAL ACTION ALTERNATIVES FOR MUNICIPAL SOLID WASTE LANDFILL SITES

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ABSTRACT

There are numerous remedial action alternatives (remedial measures) for use during or after closure of landfills and dumps which do not meet current environmental standards. Some of the alternatives are passive, that is, they require little or no maintenance once emplaced. Others are active and require a continuing input of manpower or electricity.

Most of the techniques discussed herein deal with the reduction or elimination of infiltration into landfills in one of five categories: surface water control, passive ground-water management, active ground-water or plume management, chemical immobilization of wastes, and excavation and reburial. The technology presented is widely used in construction but has not necessarily as yet been applied to landfill closure.

INTRODUCTION

Most of the municipal refuse in the United States is disposed of in approximately 15,000 sites, two-thirds of which are open dumps. The Solid Waste Disposal Act as amended by the Resource Conservation and Recovery Act (RCRA) of 1976 requires the phasing out of open dumps within the next 5 years. With the implementation of RCRA and more stringent State solid waste disposal practices it is anticipated that the majority of the approximately 10,000 open dumps will be closed within the next 5 years and increased volumes of solid waste will be disposed of in existing permitted sanitary landfills.

Closing open dumps and upgrading existing landfills both involve minimizing or eliminating the potential for resource contamination. Remedial measures are used to reduce the amount or concentration of the contaminants produced and/or to prevent or redirect their movement from the disposal site. The procedures described here have been generalized and although widely used by the construction industry, they have not been applied to landfill closure. For some waste disposal sites remedial methods may be suitable, while for other sites, because of their size or speficic characteristics, these methods may be impractical.

REMEDIAL ACTION ALTERNATIVES

To illustrate the remedial methods, a hypothetical 4-hectare (10-acre) landfill located in the northeastern United States in a depression situated on a slope between an upland ground-water recharge area and a ground-water discharge area with a stream is utilized. The landfill is underlain by 30 m (100 ft) of unconsolidated, fairly permeable materials of either glacial, coastal, or saprolitic origin, underlain by an indeterminate bedrock. The water table is 6 m (20 ft) below the ground surface and the lower 3 to 5 m (10 to 15 ft) of the landfill is in the ground water.

The landfill originated as a pushover burning dump. The waste is 12 to 15 m (40 to 50 ft) deep and extends 3 to 5 m (10 to 15 ft) above the ground surface. The bottom 3 to 5 m (10 to 15 ft) was dumped rather than landfilled and is located below the water table. The remaining waste was landfilled and covered with very sandy material. Mainly municipal refuse has been accepted at the site. However, during the time it was operated as a dump, no records of the materials deposited were kept, and it is possible that several nearby industries may have dumped industrial and/or hazardous wastes into the landfill.

To minimize pollution from a solid waste disposal site, leachate generation and movement must be limited or controlled. Leachate production can be controlled by minimizing the amount of water entering the landfill. Reducing the amount of water in contact with the fill reduces the quantity of leachate generated. However, if the quantity of water is not sufficiently minimized, the pollutional load may not be reduced at all and, indeed, may be increased due to higher contaminant concentrations. Therefore, if water inflow reduction is to be used as a remedial measure, the reduction must be significant.

It is generally not possible to reduce or eliminate the amount of moisture present in municipal refuse when collected. However, both vertical and horizontal percolation into the fill can be controlled by a number of techniques designed to either increase or decrease flow. The techniques considered herein have been grouped in five categories: surface water control, ground-water control, plume management, chemical immobilization, and excavation and reburial. It may be necessary to apply more than one method from more than one category to effect significant results. For ease of comparison, the major characteristics and estimated costs based on the hypothetical 4-hectare (10-acre) landfill of these alternatives are summarized in Table 1.

SURFACE WATER CONTROL

Surface water infiltration can often be reduced during normal landfill closure if careful design and construction practices are followed. There are four ways to minimize water infiltration. The first is to increase runoff from the landfill surface by regrading to provide for moderate sheet flow from the surface. The second is to reduce the amount of runoff flowing onto the landfill surface by constructing diversion ditches and terraces. The third is to decrease infiltration into the landfill by applying a low permeability cover or seal to retard the vertical movement of water below the cover material. The fourth is to increase interception and transpiration of precipitation by planting vegetation on the landfill.

To increase runoff from the landfill by regrading (see Figure 1), the surface should be sloped such that the water has the shortest possible flow path from any point on the site. Thus a mound in the central portion sloping equally on all sides is an ideal regrading plan. Depending on the type of soil on the site, slopes of 6 to 12 percent are generally recommended. The cost of this regrading will depend upon the current grade of the landfill and the availability

TABLE 1. SUMMARY OF ESTIMATED COSTS AND CHARACTERISTICS OF REMEDIAL METHODS

Method	Average Estimated Costs* ($ in Thousands)	Characteristics/Remarks
		Surface Water Control
Contour Grading	184	Increases runoff, reduces infiltration.
Surface Water Diversion	20	Diverts surface water from fill.
Surface Sealing		
Clay [15-46 cm (6-18 in.)]	234	If locally available, native clay is economical means of retarding infiltration.
Bituminous Concrete [4-13 cm (1.5-5 in.)]	315	Rapid coverage; can eliminate infiltration.
Fly Ash [30-60 cm (12-24 in.)]	235	Material may leach metals; may be available free.
PVC (30 mil)	482	Very impermeable; expensive seal; careful subgrade preparation is necessary.
Drainage Field (if required)	65	Carries infiltrated water off seal; increases effectiveness of seal.
Revegetation on Slopes<12 percent	10	Stabilizes cover material; seasonally increases transpiration; provides aesthetic benefit.
on Slopes>12 percent	19	
		Groundwater Control
Bentonite Slurry Trench	670	Simple construction methods; retards groundwater flow.
Grout Curtain	1,400	Very effective in permeable soils.
Sheet Piling	800	Widely used for shoring.
Bottom Sealing	4,000	Leachate collection may be needed; acts as a liner; difficult drilling through refuse.
		Plume Management**
Drains	23	Effective in lowering water table a few meters in unconsolidated materials; can be used to collect shallow leachate.
Well Point Dewatering	185	Suction lift limits depth to 7-9 m (20-30 ft); inexpensive installation; uses only one pump; can be used to collect shallow leachate.
Deep Well Dewatering	183	Used in lowering deep water tables; one pump needed per well; high maintenance costs.
Injection/Extraction Barrier	199	Creates a hydraulic barrier to stop leachate movement; operation and maintenance costs are high.
Spray Irrigation	336	Spreads leachate over the landfill for recycling; potential odor problem.
At-grade Irrigation	32	Gated pipe with ridge and furrow irrigation; potential odor problem; recycles leachate.
Subgrade Irrigation	28	Large-scale drainage field; recycles leachate.
		Chemical Immobilization
Chemical Fixation of Cover	145	Uses chemically fixed sludge to provide a top seal; provides means of disposal for sludge; helps stabilize landfill.
Chemical Injection	86	Immobilizes a single pollutant; in most cases not feasible.
		Excavation and Reburial
Excavation and Reburial	4,570	Very expensive; difficult construction.

*Costs for hypothetical 4-hectare (10-acre) landfill

**Costs include present worth of 20 years, operation, maintenance, and, where applicable, power for 4-hectare (10-acre) landfill.

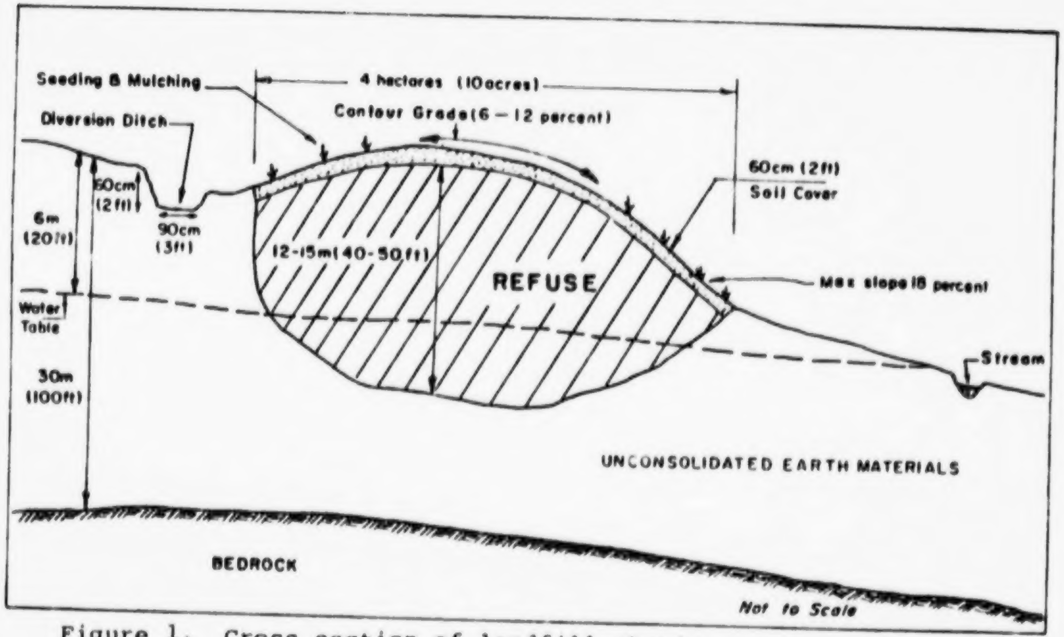

Seeding & Mulching
Diversion Ditch
4 hectares (10acres)
Contour Grade(6 – 12 percent)
60cm (2ft)
90cm (3ft)
6m (20ft)
Water Table
30m (100ft)
12-15m(40-50ft)
REFUSE
60cm (2ft) Soil Cover
Max slope18 percent
Stream
UNCONSOLIDATED EARTH MATERIALS
BEDROCK
Not to Scale

Figure 1. Cross section of landfill showing contour grading and surface water diversion.

of local cover material. If major changes in grade are necessary, costs will escalate rapidly.

Diversion ditches are generally most useful in areas where the landfill is at the middle or the bottom of a slope and surface water collects upslope from the landfill and flows onto it. Standard construction techniques developed for handling storm runoff in highway and subdivision construction can be used to redirect surface water around a landfill. These techniques are generally not excessive in cost if the equipment used for regrading and covering operations is already available at the landfill.

If the material available for covering the site is highly permeable and if the landfill is in an area of high rainfall, surface sealing may be effective (see Figure 2). A number of materials potentially suitable for sealing, in order of generally increasing costs, are local clay (where available), soil cement, bentonite, bituminous concrete, asphalt, and plastic (PVC) membranes. These materials can markedly reduce infiltration into the landfill. In cases where low permeability cover material is to be placed over the seal, it may be necessary to construct a drain on top of the seal to carry away water intercepted by it. A properly constructed seal can reduce infiltration essentially 100 percent.

An important consideration in landfill surface sealing is gas venting. Some kind of opening such as a gravel trench with mushroom cap vent (shown on Figure 2) or gas venting wells must be provided to allow escape for the gases formed by decomposing refuse. The gas can be collected in trenches or wells and, if sufficient quantities are available, it can be used or sold.

Revegetation of the completed landfill surface is recommended in favorable climates to stabilize the final cover material, aesthetically upgrade the area, and seasonally increase evapotranspiration of precipitation. Procedures for vegetating the landfill surface are very similar to those used in stabilizing highway grades and other areas of recent construction. The surface can be hydroseeded with a suitable grass mixture and mulched with a straw mulch. Steep slopes can be treated with legumes or vetch to hasten stabilization. Overland runoff over unvegetated soils rapidly erodes most cover materials and destroys the final grade. Therefore, vegetation is strongly recommended in most areas.

GROUND-WATER CONTROL

Subsurface infiltration barriers, or passive ground-water control measures, are designed to either prevent ground water from flowing through the landfill and generating leachate or control the movement of leachate away from the landfill. Barriers are constructed of low permeability materials to divert and impede ground-water flow in the vicinity of the landfill. The engineering technology associated with ground-water barriers has been developed for use in constructing cutoff walls around and under dams and excavation control structures in areas of shallow ground water, e.g., at the sea coast. Barriers that can be used at landfills include slurry trenches, grout curtains, sheet pilings, and landfill bottom sealing.

Slurry trenches (see Figure 3) are constructed by excavating a trench through a bentonite slurry using draglines. The bentonite slurry is continuously pumped into the excavation and

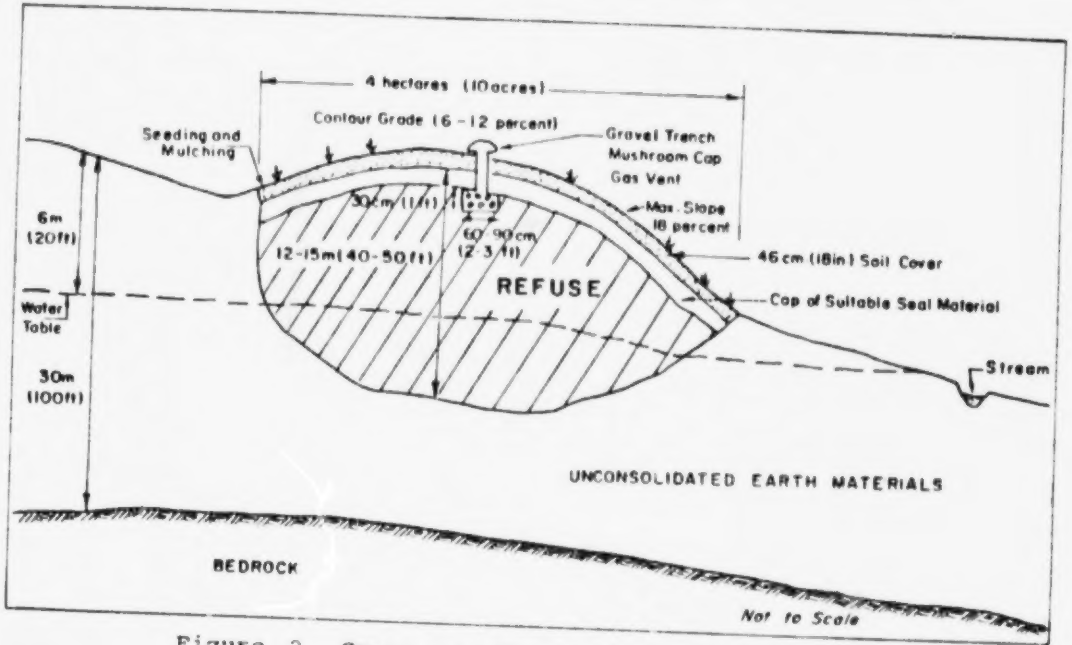
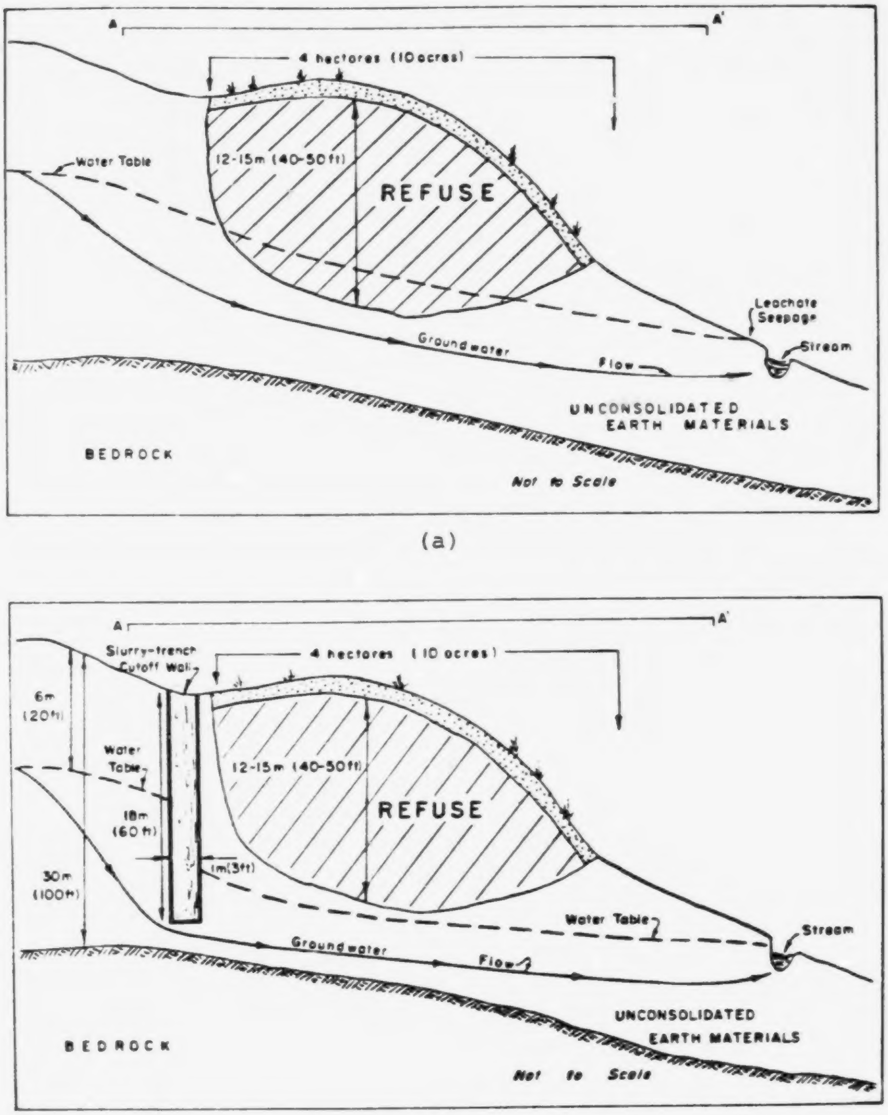

4 hectares (10 acres)
Contour Grade (6 - 12 percent)
Seeding and Mulching
Gravel Trench
Mushroom Cap
Gas Vent
Max. Slope 18 percent
46 cm (18in) Soil Cover
Cap of Suitable Seal Material
30cm (1 ft)
60-90 cm (2-3 ft)
12-15m (40-50 ft)
REFUSE
6m (20ft)
Water Table
30m (100ft)
Stream
UNCONSOLIDATED EARTH MATERIALS
BEDROCK
Not to Scale

Figure 2. Cross section of capped landfill.


A
A'
4 hectares (10 acres)
Water Table
12-15m (40-50 ft)
REFUSE
Leachate Seepage
Stream
Groundwater
Flow
UNCONSOLIDATED EARTH MATERIALS
BEDROCK
Not to Scale
A
A'
Slurry-trench Cutoff Wall
4 hectares (10 acres)
6m (20ft)
Water Table
12-15m (40-50ft)
18m (60 ft)
REFUSE
1m(3ft)
30m (100ft)
Water Table
Stream
Groundwater
Flow
UNCONSOLIDATED
EARTH MATERIALS
BEDROCK
Not to Scale

(a)

(b)

Figure 3. Cross section of landfill before (a) and after (b) slurry-trench cutoff wall installation.

serves to stabilize the nearly vertical wall of the trench and ultimately to seal the area. The excavation is often carried to bedrock or other low permeability layer but can be terminated at shallower depths. Although costly, slurry trenches are generally the least expensive form of passive ground-water barrier.

Grout curtains (see Figure 4) are emplaced by forcing a thin cement grout through tubes which are driven deep into the ground on 2- to 3-foot centers and withdrawn slowly. Two or more rows of grout are usually needed to provide a seal. Like a slurry trench, the grout is generally emplaced down to an impermeable layer. Grout curtains are quite expensive when constructed to the dimensions necessary to cut off ground-water flow in the vicinity of a reasonably sized landfill.

Sheet piling (see Figure 5) has been extensively used in near-shore and offshore construction to stabilize areas for excavation. It is driven into the ground with a pile driver so that depending on the design the piles either butt or interlock. For sheet piling, as for all of the passive groundwater barriers, unconsolidated material must be present around the landfill with no large stones or boulders as these will deflect or prevent the piling from being effectively driven. Sheet piling is generally very expensive when used in the quantities necessary at landfills.

Bottom sealing (see Figure 6) of a landfill is like emplacing a grout curtain in a bowl shape under the landfill. Grout tubes are driven through the landfill, which can entail considerable expense because of bulky or resistant refuse. A bottom-grouted landfill is similar to an engineered landfill with a liner except that the seal is emplaced after filling. Generally speaking, unless the sources of leachate generation were removed, leachate collection would be necessary with a bottom-grouted landfill.

PLUME MANAGEMENT

The purpose of plume management or active ground-water management is to manipulate the water table in the area of the landfill to either prevent the formation of leachate or contain its spread. To do this, water must be extracted from or added to the ground-water system through drains, shallow well points, or deep wells. The capital costs of these systems are generally much lower than those of passive barriers, but the operation and maintenance costs are considerably higher since in most cases a continuing supply of electrical power and ongoing maintenance of pumps and wells is required.

The technology of drains has been developed for use in agriculture, highway, and construction. Drains (see Figure 7) can be a low-cost means of lowering the water table a few meters provided the area in which the drain is to be emplaced can be readily excavated. Drains can be used to lower the water table upgradient of the fill, to collect leachate if it is following a shallow flow path, or to introduce leachate over the refuse on top of the fill. Drains are generally constructed using crushed stone and perforated pipe. Construction costs are comparatively low providing the depth of drain emplacement is not excessive. However, when highly mineralized waters are present, drains are more susceptible to clogging and maintenance costs may be significant.

Well points (see Figure 8) are widely used in construction

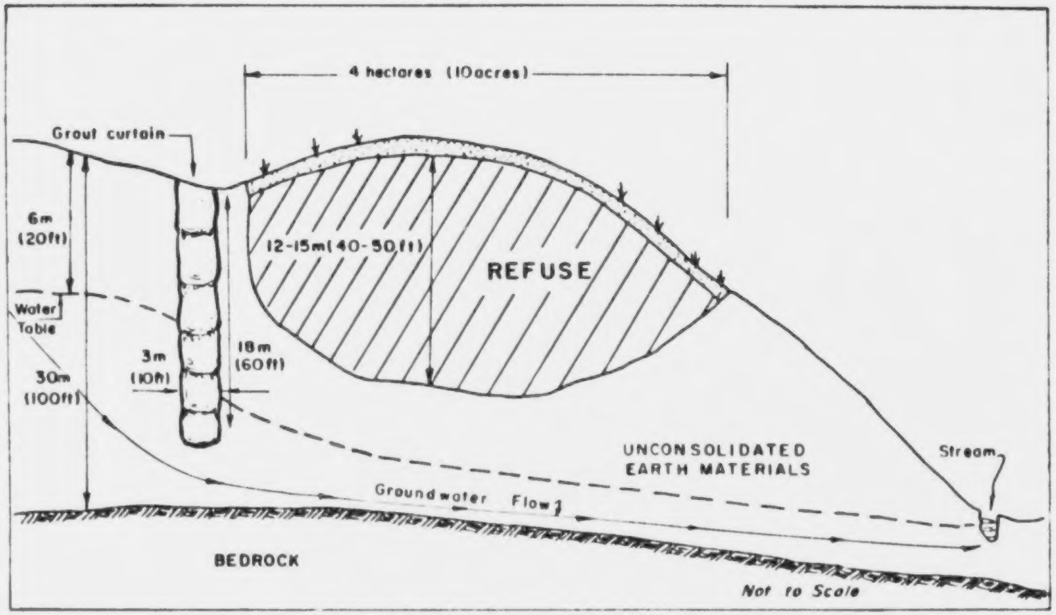

4 hectares (10acres)
Grout curtain
6m (20ft)
Water Table
30m (100ft)
3m (10ft)
18m (60ft)
12-15m(40-50ft)
REFUSE
UNCONSOLIDATED EARTH MATERIALS
Stream
Groundwater Flow
BEDROCK
Not to Scale

Figure 4. Cross section showing a grout curtain at landfill.

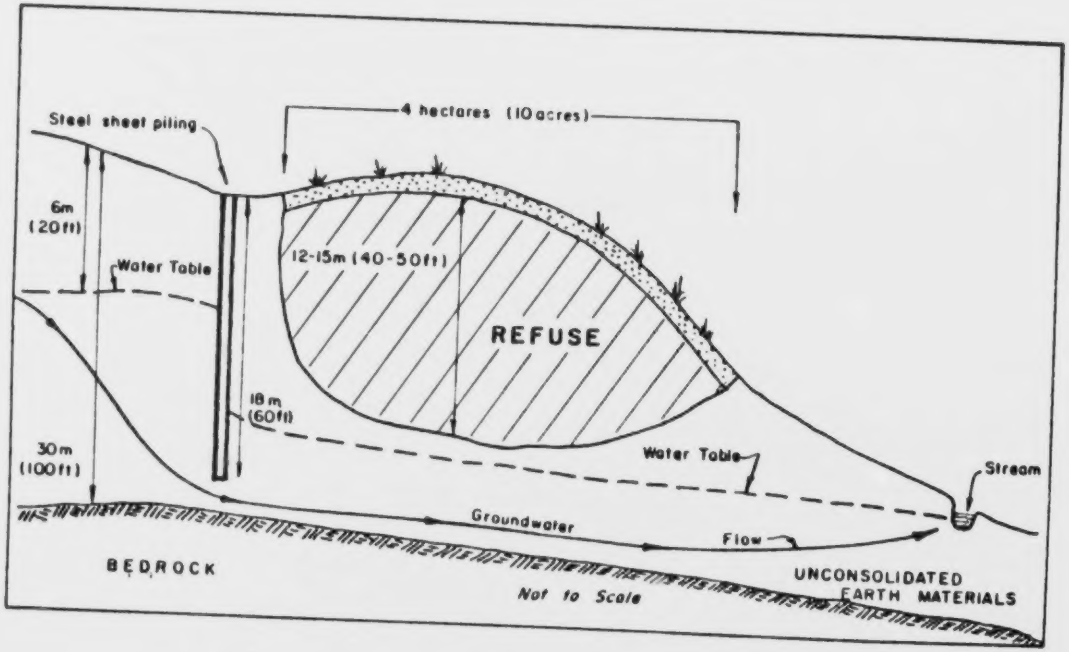

Steel sheet piling
4 hectares (10 acres)
6m (20ft)
Water Table
12-15m (40-50ft)
REFUSE
18m (60ft)
30m (100ft)
Water Table
Stream
Groundwater
Flow
BEDROCK
UNCONSOLIDATED EARTH MATERIALS
Not to Scale

Figure 5. Cross section showing a sheet piling cutoff wall at landfill

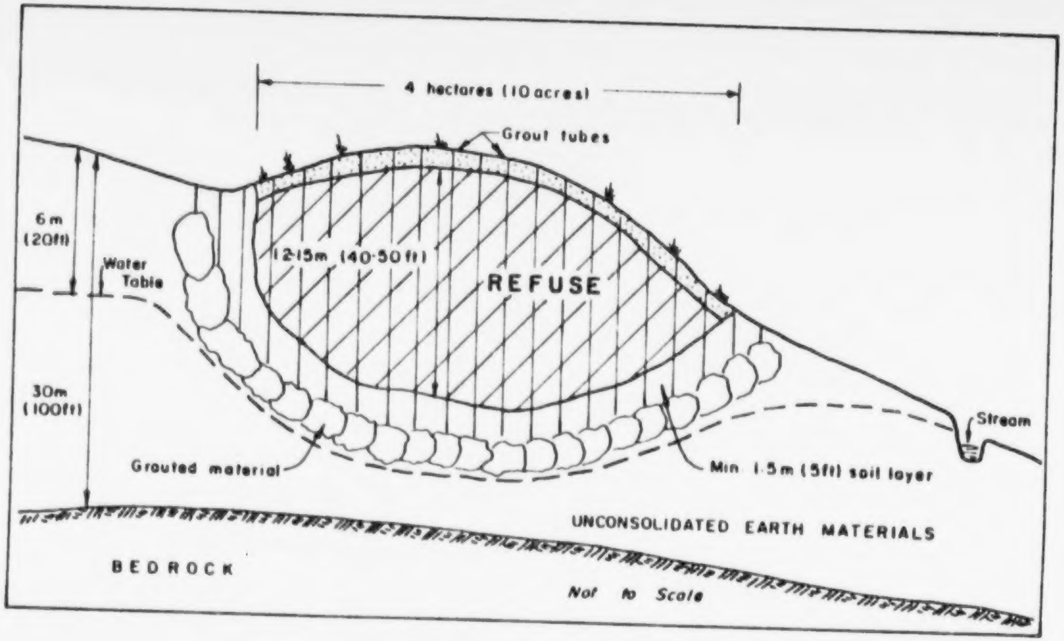

4 hectares (10 acres)
Grout tubes
6m (20ft)
Water Table
12-15m (40-50ft)
REFUSE
30m (100ft)
Stream
Grouted material
Min 1.5m (5ft) soil layer
UNCONSOLIDATED EARTH MATERIALS
BEDROCK
Not to Scale

Figure 6. Cross section showing a grouted bottom seal beneath landfill.

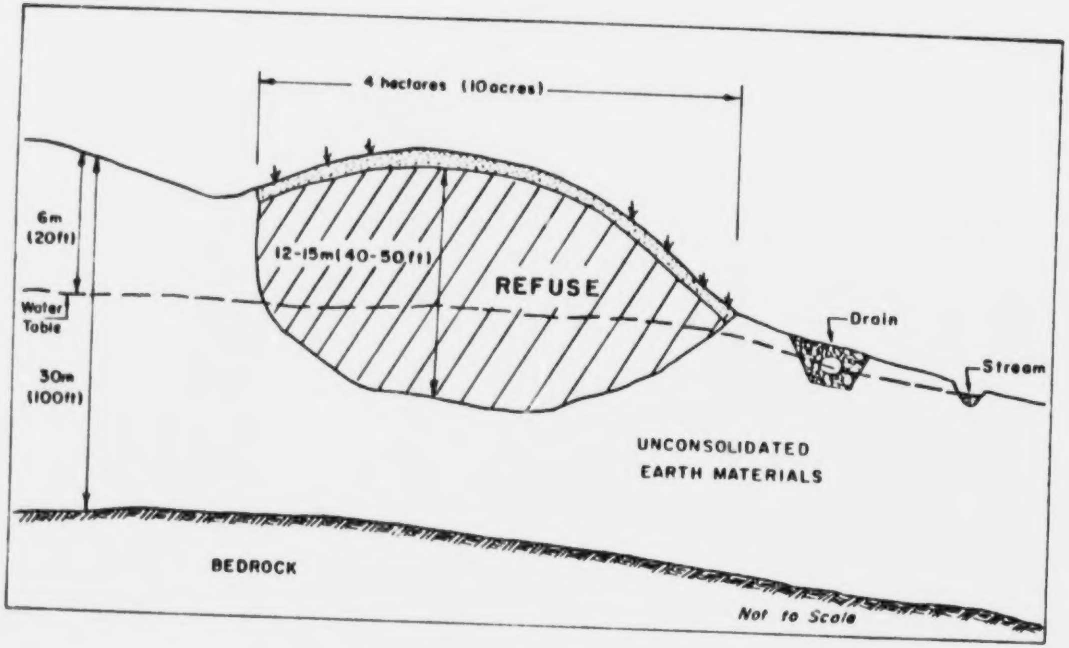

4 hectares (10 acres)
6m (20ft)
Water Table
30m (100ft)
12-15m (40-50ft)
REFUSE
Drain
Stream
UNCONSOLIDATED EARTH MATERIALS
BEDROCK
Not to Scale

Figure 7. Cross section showing a drain downgradient from the landfill.

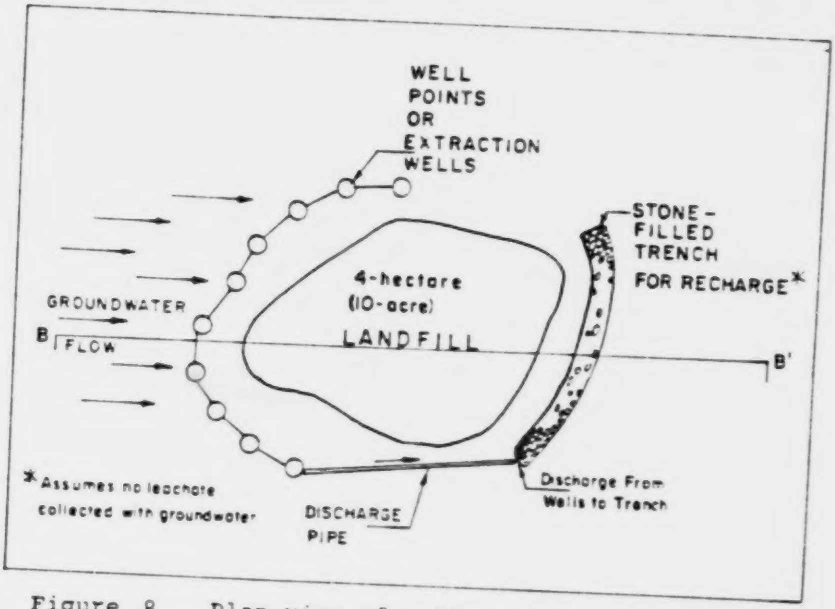

WELL POINTS OR EXTRACTION WELLS
STONE-FILLED TRENCH FOR RECHARGE*
4-hectare (10-acre) LANDFILL
GROUNDWATER FLOW
B
B'
*Assumes no leachate collected with groundwater
DISCHARGE PIPE
Discharge From Wells to Trench

Figure 8. Plan view of well points or extraction wells used to lower the water table upgradient from the landfill.

to dewater shallow excavations. They are effective up to the limits of suction lift, i.e., 7 to 9 m (20 to 30 ft) below the ground. The main advantage of well points is that a large number of wells can be powered using a single suction pump. They are effective for dewatering shallow landfills and collecting shallow leachate. The installation cost for well points is moderate but maintenance can be relatively high since a good vacuum must be maintained on the entire system.

Deep wells (see Figure 9) can be used to dewater consolidated formations or areas where the water table is too deep for economical use of suction lifts or drains. Construction and operation of deep wells is a relatively simple but long-term operation, and maintenance costs can be high especially if the wells are used to collect leachate.

Any of these systems can be used to inject water into an aquifer to provide a ground-water barrier to the flow of leachate. This technology has been developed and applied in controlling the spread of sea water into potable aquifers and is potentially applicable to control leachate movement toward important well fields. However, the operation and maintenance costs are high and the leachate is only rerouted from its path, the quantity generated is not reduced.

The use of any of these systems for leachate collection will necessitate some means of leachate disposal. It is often feasible to recycle the leachate onto the landfill to hasten landfill stabilization. Leachate can be recycled by spray irrigation, at-grade irrigation, or subgrade irrigation (i.e., drains and tile fields). Spray irrigation involves the highest capital, energy, and maintenance costs, but also provides some leachate treatment and recycling. At-grade irrigation considerably reduces the power requirements but shares with spray irrigation potential odor problems. Subgrade irrigation provides little direct leachate renovation but avoids the potential problem of having large quantities of leachate exposed at the ground surface. Generally speaking, landfills will stabilize more rapidly with leachate recycling. Potentially, a leachate collection system could be abandoned when the landfill has stabilized.

All of these ground-water management schemes involve a long-term commitment of manpower and funds to the maintenance and operation of the systems. Probably the least operation and maintenance costs would be required using drains and the most using a leachate collection and recycling system. However, there may be cases where any or all of these measures will be applicable and necessary for leachate control.

CHEMICAL IMMOBILIZATION

Chemical immobilization is a developing technology used to stabilize the waste and/or cover material through a chemical reaction. The method involves either the emplacement of a chemically stabilized low permeability cover or the injection of a chemical into the refuse and leachate plume to destroy a contaminant. The technology for chemical immobilization originated in chemical engineering and sludge stabilization work.

The first use of chemical immobilization (see Figure 10) is identical in intent to top sealing but uses a chemically stabilized waste product, i.e., sludge, to form a low perme-

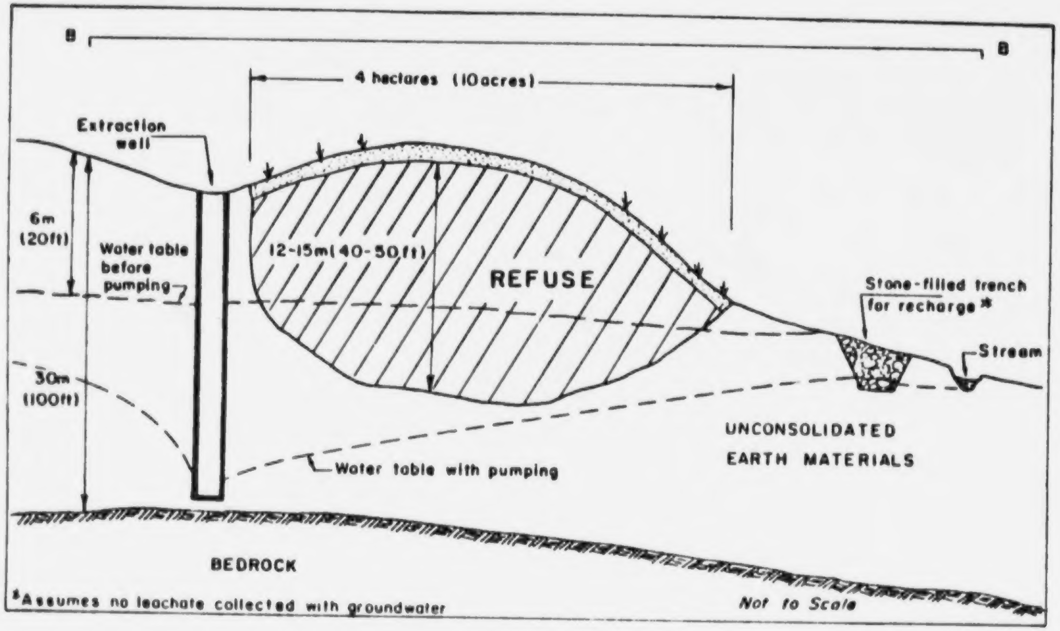

B
B
4 hectares (10 acres)
Extraction well
6m (20ft)
Water table before pumping
12-15m (40-50 ft)
REFUSE
Stone-filled trench for recharge*
Stream
30m (100ft)
UNCONSOLIDATED EARTH MATERIALS
Water table with pumping
BEDROCK
*Assumes no leachate collected with groundwater
Not to Scale

Figure 9. Cross section showing an extraction well at landfill.

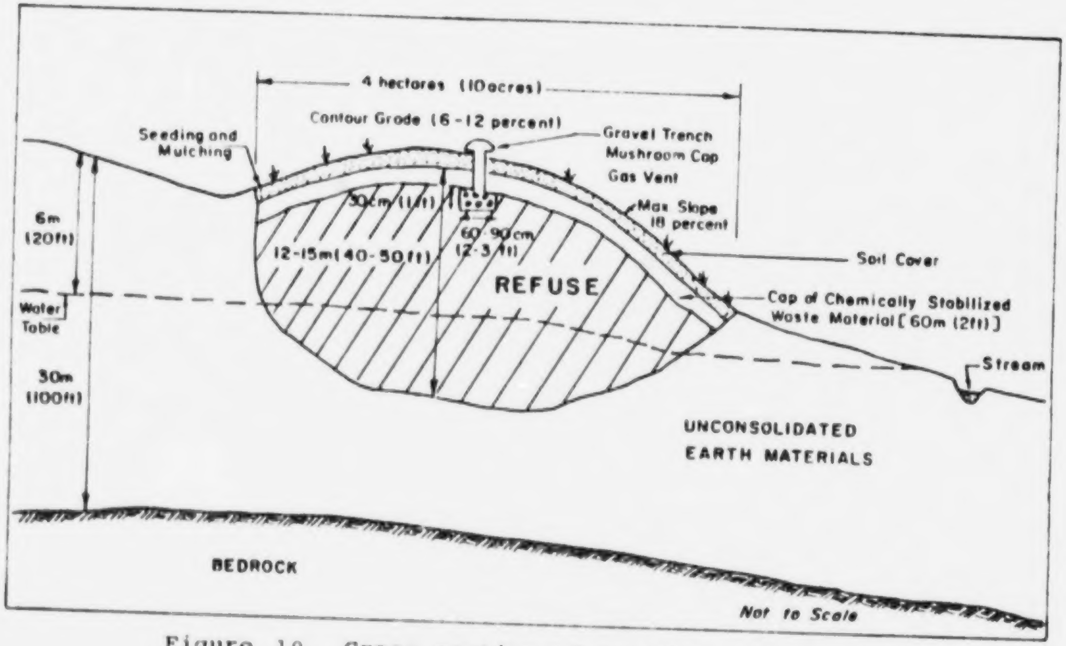

4 hectares (10acres)
Contour Grade (6-12 percent)
Seeding and Mulching
Gravel Trench
Mushroom Cap
Gas Vent
Max Slope 18 percent
Soil Cover
Cap of Chemically Stabilized Waste Material [60m (2ft)]
30cm (1ft)
60-90cm (2-3 ft)
12-15m (40-50 ft)
REFUSE
6m (20ft)
Water Table
30m (100ft)
Stream
UNCONSOLIDATED EARTH MATERIALS
BEDROCK
Not to Scale

Figure 10. Cross section of landfill sealed with stabilized waste material.

ability blanket on top of the landfill. In areas where this would be permitted, this method would have the beneficial effect of disposing of a waste product and at the same time aiding in the stabilization of the landfill. Most chemical immobilization processes are proprietary, but in general they involve either a cement base or chemical reaction base process. When a suitable waste material is available within reasonable distance of the landfill, this procedure can be cost-advantageous over other top sealing technology.

The other form of chemical immobilization (see Figure 11) is the injection of a chemical to destroy or tie up a specific pollutant. Generally speaking, any one chemical is effective only against one or two types of pollutants. This process is potentially very expensive but would be applicable in areas where a known hazardous material, such as cyanide, had been emplaced and was migrating with the ground water. Chemical immobilization is a developing field and, at present, most methods of chemical immobilization would not be feasible for municipal refuse.

EXCAVATION AND REBURIAL

The purpose of excavation and reburial is to move the leachate-generating material to a better engineered or environmentally less sensitive area. Although conceptually very simple, excavation and reburial is expensive as well as practically and politically difficult. Basically, the process involves removing the entire contents of the landfill with common construction procedures and moving them, usually by truck, to another area where a better engineered and sited landfill is available. Problems arising in excavation and reburial include the technical problems of removing large quantities of bulky and partially decomposed wastes from the landfill, transporting partially saturated and saturated material over public roads without spillage or leakage, and cleaning up a leachate-filled pit after the waste material has been removed.

In addition, in most cases there are political constraints on the movement of such material through any populated areas, and there may be considerable local opposition to the importation of municipal refuse to an existing or new landfill. The process is in any case very costly, but in areas where a severe leachate problem has developed and a properly designed landfill is available nearby it may be applicable.

SUMMARY

Surface regrading and revegetation is useful and necessary at practically all landfills, and surface sealing is a useful adjunct to this where no natural low permeability cover is available. Groundwater manipulation is potentially very effective in areas where either much of the landfill is below the water table or the leachate is moving toward an important public water supply. Leachate collection and recycling may be a useful part of groundwater control. Chemical immobilization is a relatively new technology whose applicability will increase as more techniques are developed; however, the use of a stabilized sludge or other waste product for cover or sealing material appears to be feasible in areas where it would not be prohibited by local authorities and where the material is locally available. Excavation and reburial should be generally considered as a last resort which would be used only when all efforts to stabilize the refuse in place appear to be futile. Although it is poten-

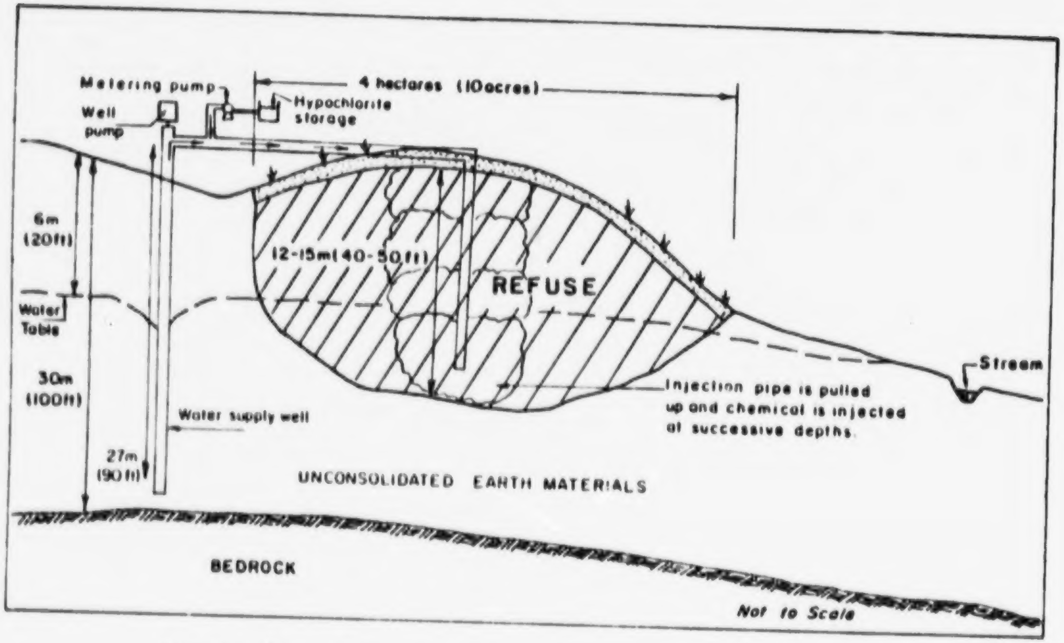

Metering pump
Well pump
Hypochlorite storage
4 hectares (10acres)
6m (20ft)
Water Table
30m (100ft)
27m (90ft)
12-15m(40-50ft)
REFUSE
Water supply well
Injection pipe is pulled up and chemical is injected at successive depths.
Stream
UNCONSOLIDATED EARTH MATERIALS
BEDROCK
Not to Scale

Figure 11. Cross section of landfill treated by chemical injection.

tially very effective it is also a difficult and expensive undertaking.

In the selection of remedial measures for use in minimizing pollution at any waste disposal site, professional feasibility assessment and design will be necessary.

ACKNOWLEDGEMENTS

This paper concerns itself with one phase of a multiphase project being conducted by A. W. Martin Associates, Inc. under U.S. EPA Contract No. 68-03-2519, Donald E. Sanning, Project Officer. Other phases involve the selection of an abandoned waste disposal site for study, the design and implementaion of remedial neutralization procedures, and the implementation of a monitoring program to determine the effectiveness of the procedures. A comprehensive discussion of remedial measures and estimates of their costs can be found in "Guidance Manual for Minimizing Pollution from Waste Disposal Sites" (EPA-600/2-27-142).

REMEDIAL ACTION ACTIVITIES FOR ARMY CREEK LANDFILL

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ABSTRACT

Groundwater contamination due to leachate emanating from the Army Creek Landfill in New Castle County, Delaware, has been controlled by a system of pumping wells. From the outset, the wells were considered to be temporary control measures until a permanent solution was developed.

After five years of extensive research, the recommended permanent control measures are still hydrogeologic in nature. They include;

- Minimize leachate production by decreasing precipitation infiltration and groundwater flow.
- Maximize leachate recovery by locating contaminant recovery wells closer to and within the landfill.

Initial costs associated with this program are approximately $4.4 million with annual operating and maintenance costs approaching $990,000 per year.

SITE HISTORY

Between 1960 and 1968, New Castle County, Delaware utilized an abandoned sand and gravel pit as the primary disposal site for municipal and industrial wastes. The landfill location in Northern Delaware is shown on the map in Figure 1.

From records and test borings, it was determined that a modified area fill method was used to emplace the refuse material. During the landfilling operation, daily covering was probably not practiced. This, coupled with less than adequate compaction within the areas, caused differential settling and an uneven finished surface when the fill was closed.

The final lift was emplaced in 1968, and the completed fill encompassed a volume of 1.9 million cubic yards of refuse, about 30 percent (or 600,000 cubic yards) of which was beneath the seasonal high water table. Refuse thicknesses

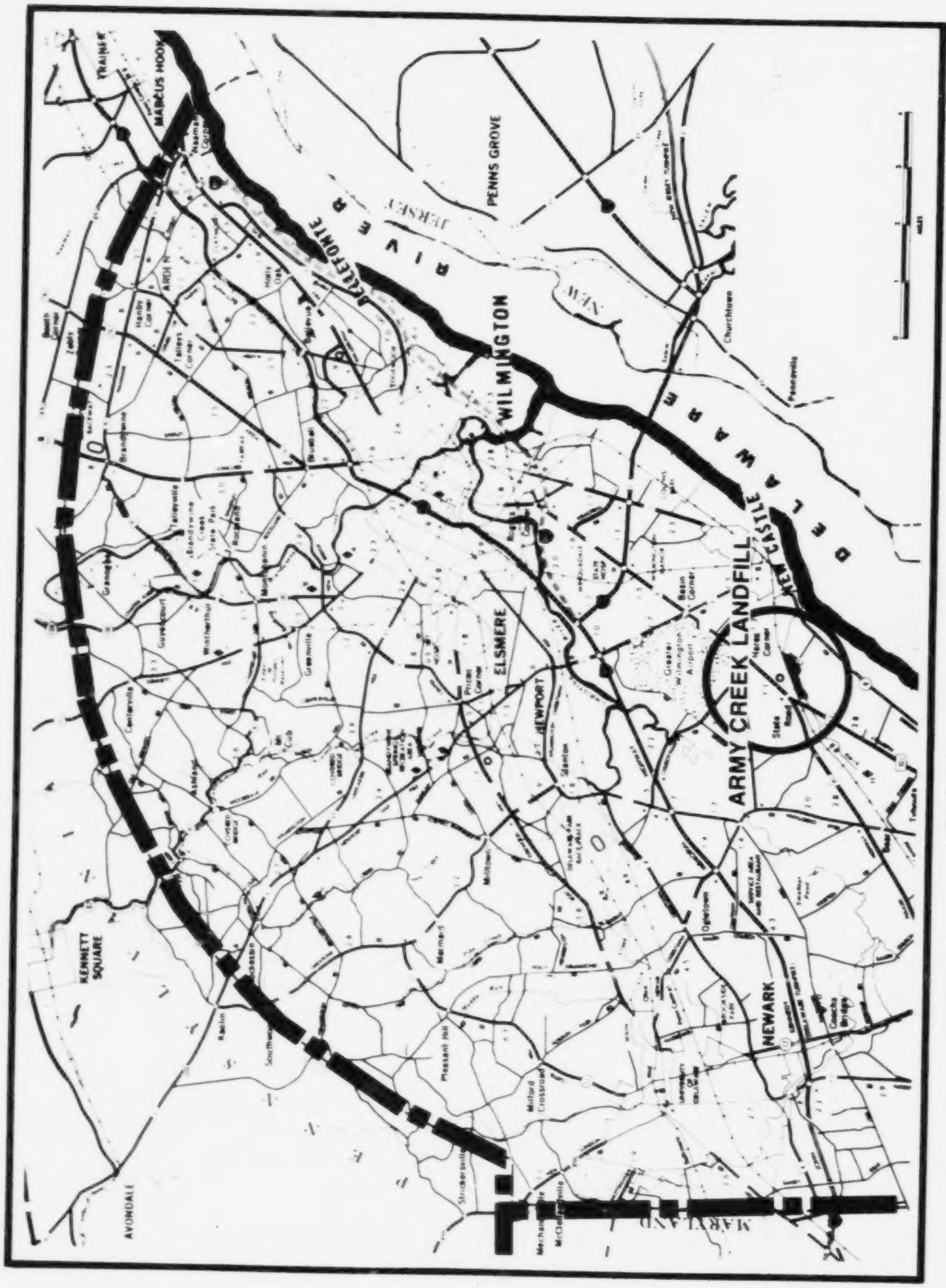

MARCUS HOOK
TRAINER
BELLEFONTE
RIVER
JERSEY
PENNS GROVE
NEW
WILMINGTON
DELAWARE
NEW CASTLE
ARMY CREEK LANDFILL
Hares Corner
State Road
Basin Corner
Greater Wilmington Airport
Churchtown
Pennsville
ARDEN
ELSMERE
NEWPORT
Stanton
Ogletown
NEWARK
Milford Crossroads
Pleasant Hill
Mechanicsville
KENNETT SQUARE
AVONDALE
MARYLAND
P E N N A

FIGURE 1 LOCATION MAP OF ARMY CREEK LANDFILL

over the 47-acre site range from about 6 to over 35 feet.

Late in 1971, water in a residential well, located 900 feet southwest of the landfill, developed quality problems such as a distinctly disagreeable odor and permanent staining of porcelain fixtures. Gradually, this condition became more pronounced and the water supply abandoned. Analyses conducted by the Delaware Geological Survey and New Castle County Public Works Department indicated the presence of leachate in the ground water. In early 1972, intensive field study was begun by New Castle County through its consultant, Roy F. Weston, Inc., and the Delaware Geological Survey to determine the hydrogeologic circumstances responsible for this problem.

HYDROGEOLOGIC DESCRIPTION

The surficial geology of the old gravel pit/landfill consists of medium and coarse-textured, channel-bedded, unconsolidated sands that demonstrate substantial textural variability within a short vertical section. The origin of these sands has been related to glacial meltwaters having seasonal discharges. The deposits settled in shallow, relatively narrow braided channels with numerous ice floes. These deposits have been formally named the Columbia Formation and within the study area form a continuous surficial layer up to 60 feet in thickness with the formational base varying between +20 to -20 feet present mean sea level. Dragline excavations in the gravel pit during quarrying had terminated at a basal quartz conglomerate It has been determined from air photos that in at least two places within the gravel pit this basal conglomerate was removed by excavation and the bottom of the gravel pit was continued into the underlying aquifer sands of the early Cretaceous Potomac Formation which has been heavily developed for ground water since the early 1960's.

In the immediate area, well pumpage has exceeded 8 mgd during this decade. Consequently, the natural southerly ground water gradient has been steepened, and the rate of ground water movement has been greatly accelerated. This was an alarming discovery considering a wellfield belonging to a major water company lies 3,000 feet southwest of the landfill. See Figure 2.

CONTAMINATION HISTORY

By mid-1972, an investigation had been undertaken to determine the extent of the leachate-affected area. Figure 2 shows the leachate plume as determined by data gathered from an expanded monitor well network. This line is an approximate location based upon specific parameters used as standard indicators. These parameters included physical appearance of water, e.g. color and odor, and COD, total iron, manganese, and chlorides. The concentration of iron and manganese in aquifer waters is such that they are useful as quality parameters. Their gradual increase over background conditions has, therefore, indicated: (1) remobilized metals under nonequilibrium conditions and/or (2) fresh dissolved metals carried in solution. The COD levels within the landfill were normally high (see Table 1). A few hundred feet south in monitoring wells nearest the landfill the COD levels had dropped significantly, indicating that organic decomposition of the landfill was subsiding or, more

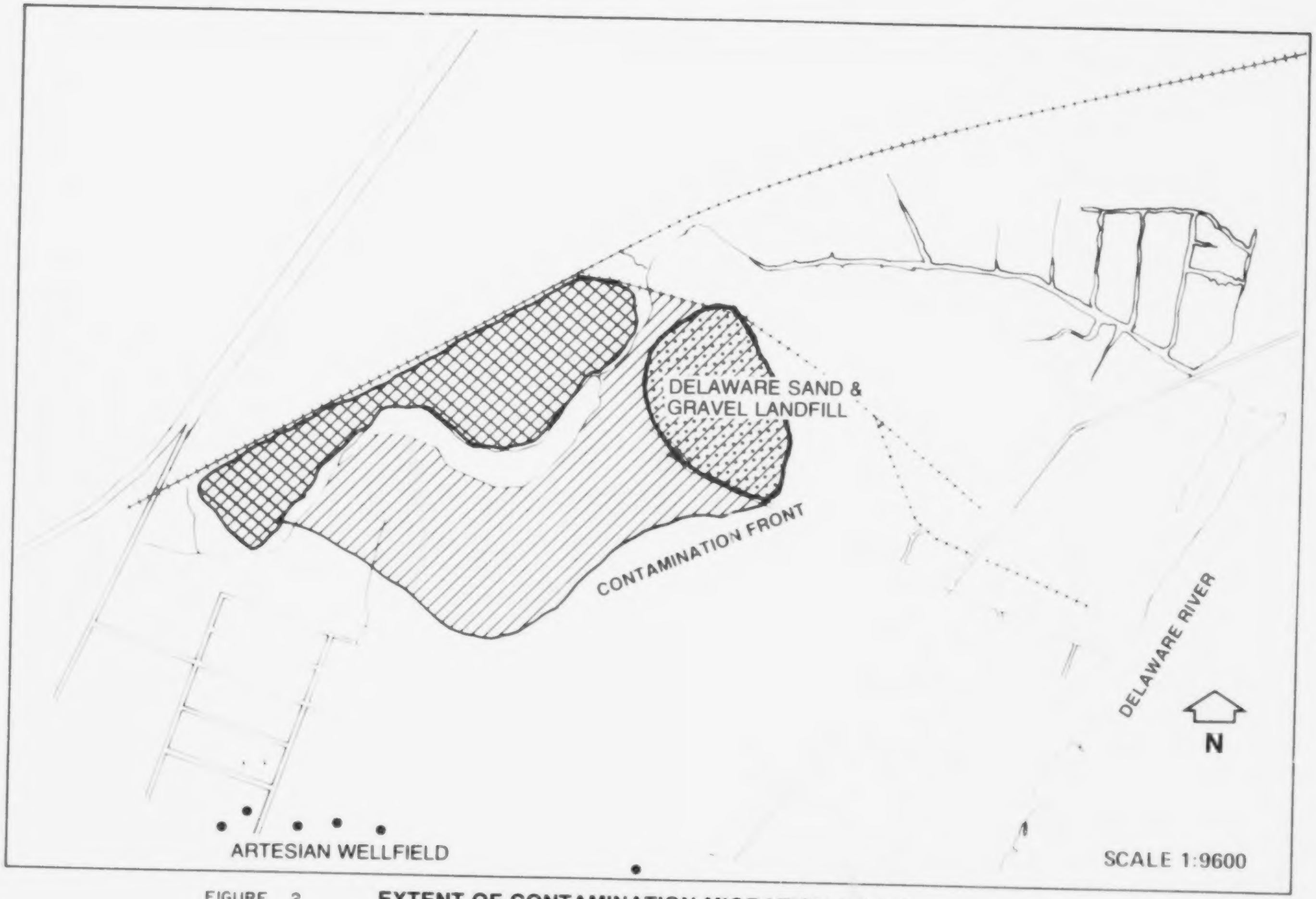

DELAWARE SAND &
GRAVEL LANDFILL
CONTAMINATION FRONT
DELAWARE RIVER
N
ARTESIAN WELLFIELD
SCALE 1:9600

FIGURE 2 **EXTENT OF CONTAMINATION MIGRATION AS OF AUGUST 1973 IN RELATION TO THE DELAWARE SAND & GRAVEL LANDFILL AND THE ARTESIAN WATER COMPANY WELLFIELD**

likely, that fresh, oxygenated recharge waters were fully oxidizing the organics within the Potomac aquifer through interformational leakage.

Based on aquifer characteristics of the Upper Potomac, the rate of contaminant migration was computed. The permeability of the channel sands of the Potomac is greater than 500 gpd/sq. ft. The hydraulic gradients, as can be inferred from the potentiometric surface map in Figure 3, vary from about 0.015 ft./ft. immediately south of the landfill to 0.02 ft./ft., as the gradient steepens nearer the production wells. An average rate of movement of 0.5 feet per day is reasonable for the area close to the landfill and increases to 1.5 ft./day down-gradient closer to the pumping centers of the nearby water company wells.

As leachate enters the Upper Potomac aquifer, barring attenuation of select constituents, it was expected to reach the water company's pumping wells within 10 years if no corrective procedures were undertaken.

LEACHATE QUALITY

The quality of leachate generated by saturated decomposition of domestic refuse depends upon such variables as waste composition and sorting, compaction, moisture capacity, temperature, and age of the refuse. Thus, the actual concentrations of chemical species in various leachates vary greatly. A young landfill will generally produce a leachate characterized by very high concentrations of dissolved organic and inorganic substances. Both the Chemical Oxygen Demand (COD, a measure of dissolved organics) and the Total Dissolved Solids concentration (TDS, a measure of dissolved inorganics) are typically in the thousands of parts per million for the raw leachate. The concentration of most environmentally significant metals greatly exceeds the recommended EPA limits for such substances in potable water. The leachate concentration rapidly decreases as it migrates down-gradient from the landfill by such processes as dilution, bioassimilation, chemical oxidation, and adsorption.

The background water quality in the Potomac aquifer has a low dissolved solids concentration. The range of inorganic constituents in natural Potomac waters is listed in Table 1, along with comparison values from selected observation wells located within and down-gradient of the landfill.

COUNTER-PUMPING PROGRAM

The installation of a network of monitoring wells was completed in early 1973, so that the ground water area affected by migration of leachate could be fully identified.

Through an analysis of the initial flow network in the Potomac aquifer and computed well interferences from selected observation wells, an interim contaminant control was designed. The control consists of an artificially depressed drainage divide within the Upper Potomac aquifer, down-gradient from the landfill at about a third of the linear distance to the water company wellfield. The drainage divide, as designed, was composed of a two-part well interference system. Counter-pumping wells were systematically located within the channel sands of the Upper Potomac aquifer to create the initial

Map showing the theoretical ground water flow pattern in the upper Potomac aquifer based on piezometric and stratigraphic information in the vicinity of the Army Creek Landfill, September 1972

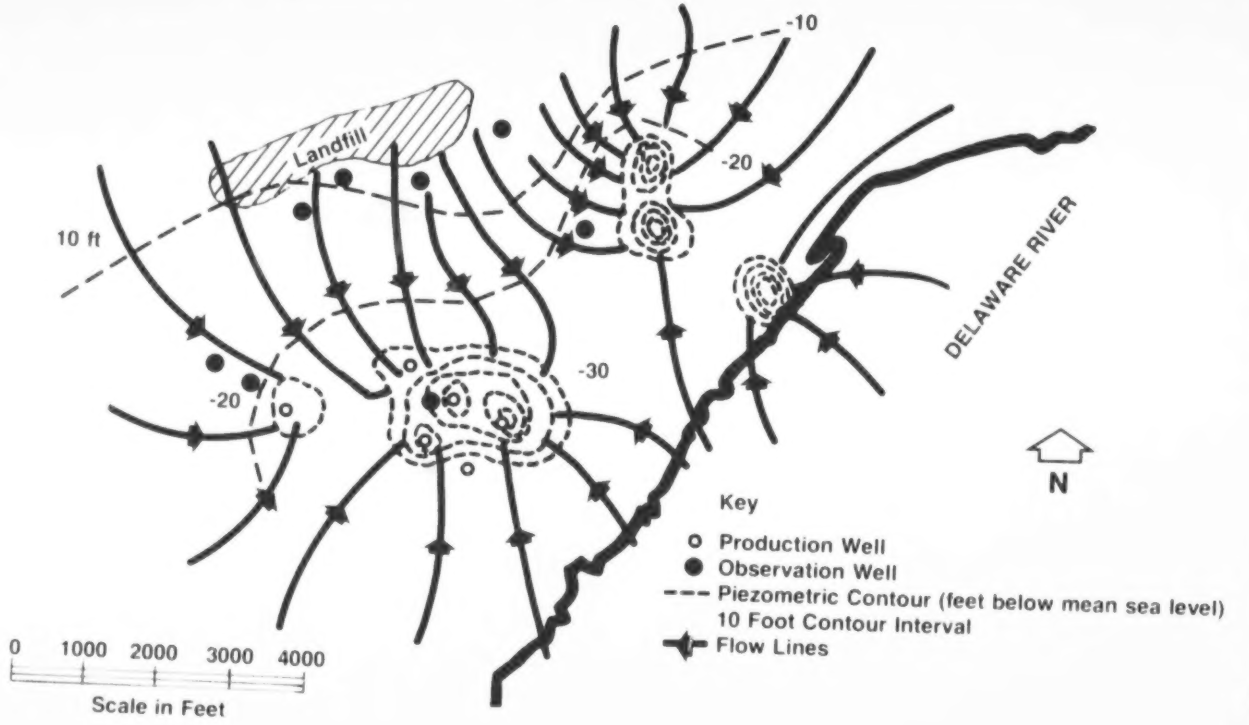

-10
Landfill
-20
10 ft
DELAWARE RIVER
-30
-20
N
Key
○ Production Well
● Observation Well
--- Piezometric Contour (feet below mean sea level)
10 Foot Contour Interval
Flow Lines
0 1000 2000 3000 4000
Scale in Feet

FIGURE 3 POTENTIOMETRIC SURFACE MAP OF THE UPPER POTOMAC AQUIFER PRIOR TO INSTALLATION OF CONTROL MEASURES

	Background Quality Upper Potomac Aquifer	Water table Beneath Landfill	Potomac Aquifer 400 ft. Downgradient from Landfill
COD (ppm)	14	5617	170
TDS (ppm)	26	3606	933
NH N (ppm)	55	113 4	72.2
NO N (ppm)	1 75	—	—
Cl (ppm)	9 2	436	236
ΣFe (ppm)	20	550	47
Phenols (ppm)	20 ppb	18	01
Sp C (μ mhos cm^{-1})	120	2500	1900

Concentration of Specific Chemicals and Contaminants Near Army Creek Landfill
TABLE 1

divide. Secondly, the water company wells were throttled back to initiate head build-up down-gradient. The first counter-pumping wells have been in operation since November 1973. At the time of completion in early 1974, the recovery wellfield was pumping a total of 3.54 million gallons of contaminated water per day.

The divide has been effective in preventing contaminated waters from flowing into the steepened gradient of the water company's wells. In plan view, the divide consists roughly of an L-shaped ground water mound south and east of the landfill as shown in Figure 4. The flow lines again indicate the preferred flow path of a particle within this network.

An added feature to the divide is the physical removal of contaminants at that point. The leakage characteristics through the Columbia Formation south of the counter-pumping network then allow fresh recharge to enter the aquifer. The recovered contaminated water is allowed to settle within a perched retention pond south of the landfill. Through mild oxidation, certain contaminants are removed and the overflow infiltrates the Columbia Formation and the Delaware Estuary.

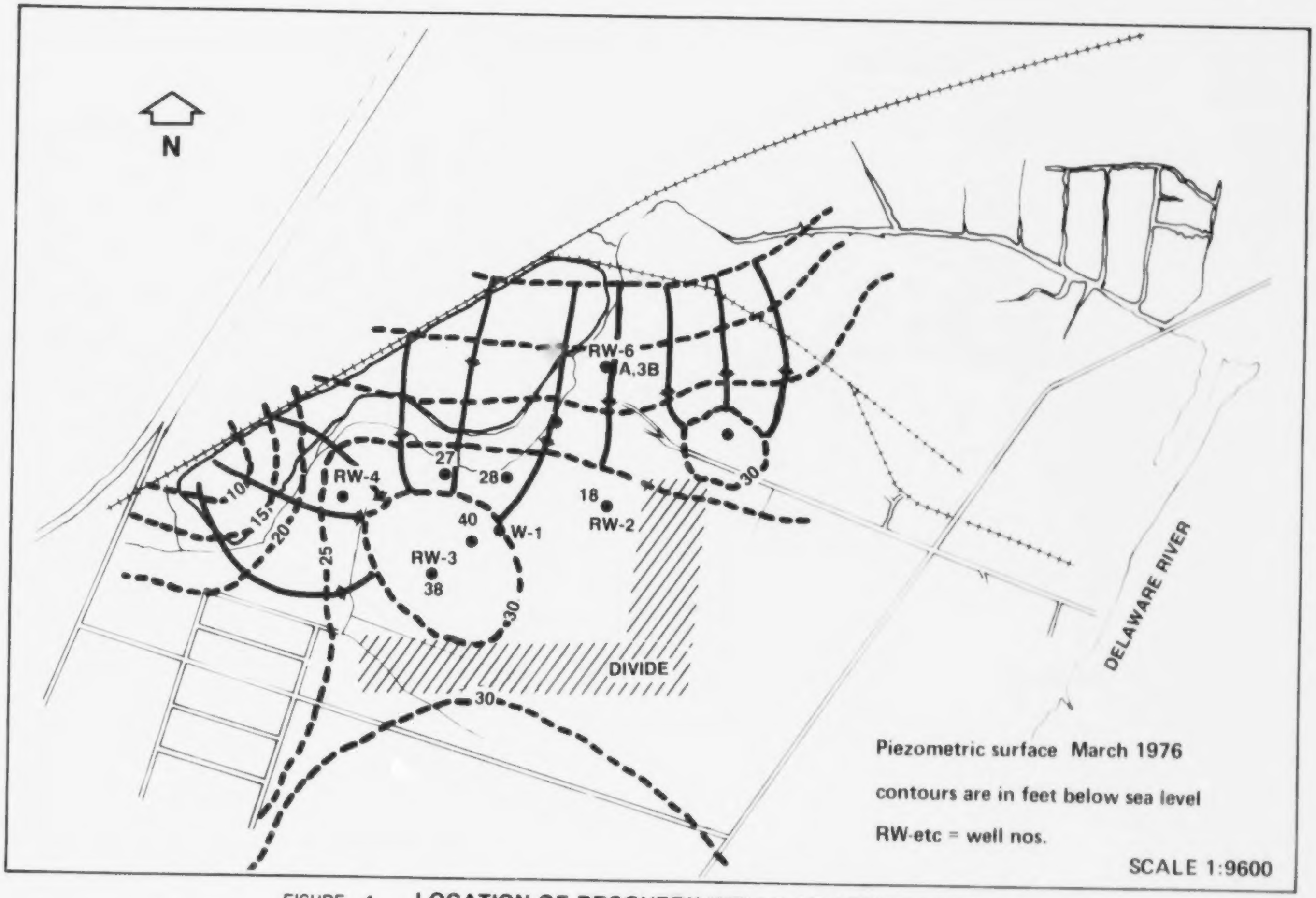
In cross section, the development of the drainage divide during the history of the leachate containment is shown in Figure 5. Line A shows the piezometric gradient from north to south between the landfill and the community wells. Line B shows the same traverse line after initiation of the control measures in late 1973. Since that time, the piezometric gradient has risen and declined with respect to increased recharge, seasonal demand, chemical and bacteriological encrustation of the recovery wells, and later rehabilitation; still it has remained fairly constant in position and effectiveness to this day.

The counter-pumping measures at the Army Creek Landfill have been successful. Figure 6 shows an interpretive line of contaminants based upon spring 1978 data. In certain fringe area monitoring points, the ground water quality has actually been shown to improve, although it is still below U.S. Public Health Service limits and EPA interim standards for potable water.

The counter-pumping program had, from its inception, been viewed as an emergency contaminant control program--one that would be utilized to prevent further migration while an ultimate solution was developed and implemented. Various "permanent" solutions are summarized below. It should be sufficient to state that the presented cost estimate reflects a "worst case" situation whereby each operation must be carried out to its fullest extent. Furthermore, the presented cost does not include control measures for the neighboring Delaware Sand and Gravel landfill.

I. ATTENUATION

This alternative involves utilization of natural processes to cleanse the aquifer. These processes include dilution, adsorption, and oxidation. It is concluded that attenuation is presently taking place in the aquifer, largely due to dilution of contaminated water by cleaner water entering the aquifer and, to a lesser extent, to


N
RW-6
A,3B
27
28
18
RW-2
RW-4
40
W-1
RW-3
38
10
15
20
25
30
30
30
30
DIVIDE
DELAWARE RIVER

Piezometric surface March 1976

contours are in feet below sea level

RW-etc = well nos.

SCALE 1:9600

FIGURE 4 LOCATION OF RECOVERY WELLS AT ARMY CREEK

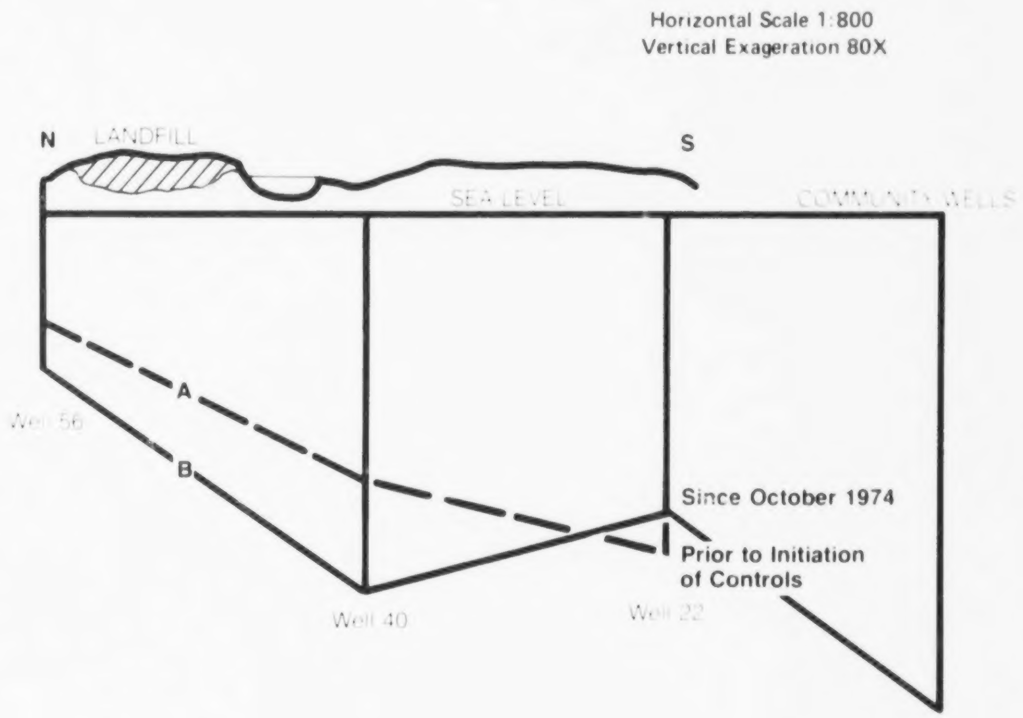

Horizontal Scale 1:800
Vertical Exageration 80X
N
LANDFILL
S
SEA LEVEL
COMMUNITY WELLS
A
B
Well 56
Well 40
Well 22
Since October 1974
Prior to Initiation of Controls

FIGURE 5 **CROSS SECTION THROUGH THE PIEZOMETRIC SURFACE "DRAINAGE DIVIDE"**

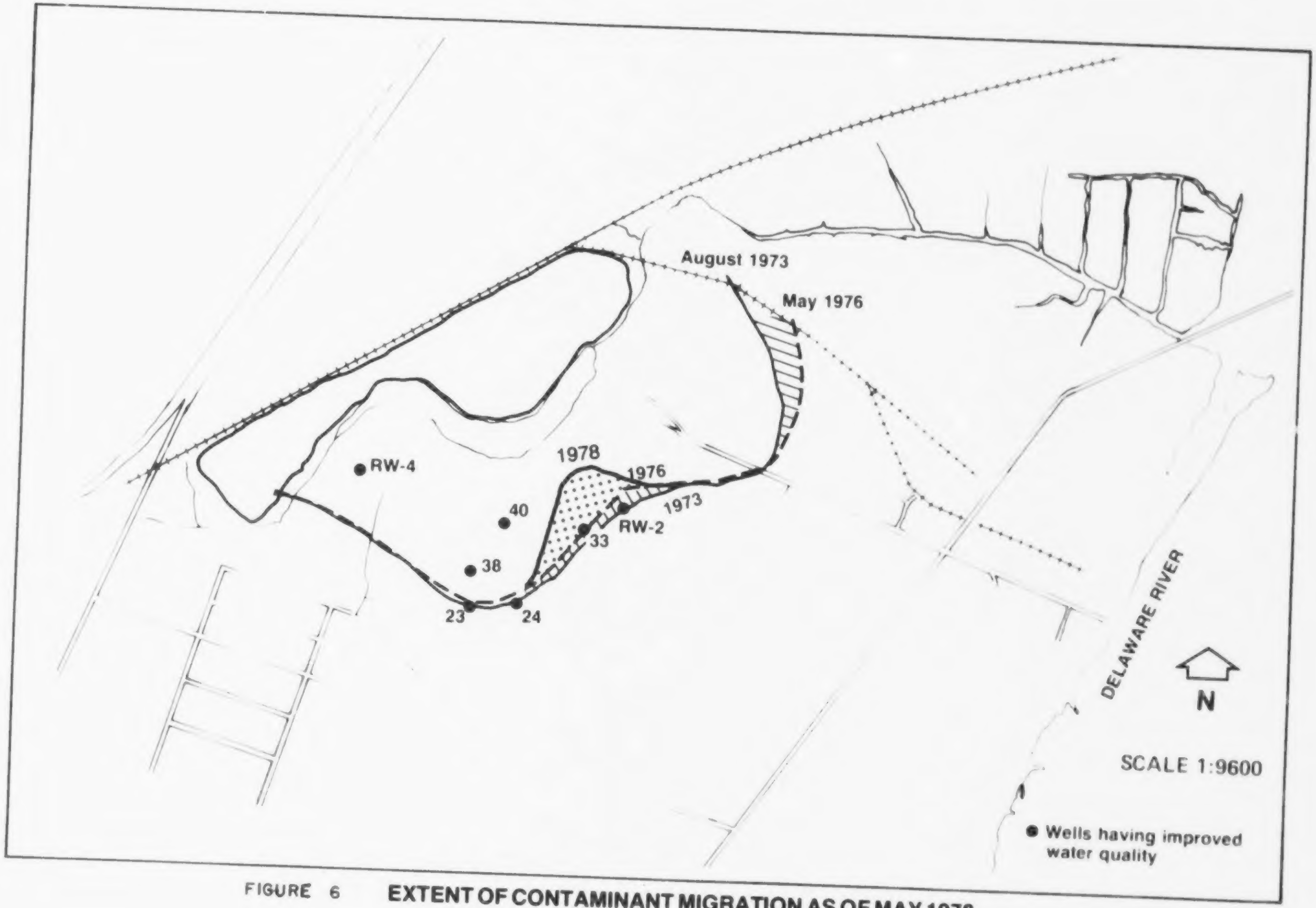

August 1973
May 1976
1978
1976
1973
RW-4
40
38
23
24
33
RW-2
DELAWARE RIVER
N
SCALE 1:9600
● Wells having improved water quality

FIGURE 6 **EXTENT OF CONTAMINANT MIGRATION AS OF MAY 1978**

adsorption of constituents to the aquifer sands. It is further concluded that the natural process of oxidation of contaminants is insignificant. Instituting this alternative could result in either (a) abandonment of the Artesian Water Company Llangollen Wellfield and development of another water supply source equal to the theorized sustained yield (4.5 mgd) or (b) treatment of the ground water at the wellhead prior to use. Since it is presently unknown whether the contaminated ground water can be treated to drinking water standards, further discussion of this alternative is limited to abandonment of the aquifer and subsequent development of another water supply source. This alternative does not necessarily imply that the aquifer is to be abandoned. Rather, it may be necessary to abandon the existing wellfield due to its proximity to the landfill or add additional treatment, i.e., carbon adsorption, to Artesian Water Company's existing treatment process to remove the organic fraction. It also may be possible, through dilution and adsorption within the aquifer, to withdraw this water at some distance downgradient for potable water supply without additional treatment. Initial capital costs for this program are approximated at $3,000,000 with yearly operation and maintenance costs of $1,232,000.

II. HYDROGEOLOGIC CONTROLS

This alternative entails controlling the water entry and exit from the landfill by the following means:

A. Precipitation Infiltration Reduction - The landfill surface may be regraded and covered with a material suitable to retard infiltration, yet conducive to vegetative cover. Approximately 80 percent of precipitation falling on the landfill surface can be prevented from infiltrating the refuse. The resultant infiltration may then average about 32,000 gpd.

B. Interception of Groundwater Inflow - Groundwater inflow to the landfill from the Pleistocene sands can be reduced by one or a combination of the following methods:

1. A well-point system
2. A series of larger diameter wells with submersible pumps.
3. A perforated tile drain.
4. A gravity-operated recharge well system screened in both the Columbia and Pleistocene sands.
5. Excavation and drainage through an open channel around the landfill.

It is anticipated that 90 percent of the present ground water inflow to the landfill could be eliminated and thus resultant inflow would be approximately 15,000 gpd.

C. Collection of Leachate Within the Landfill - Much of the leachate could be collected by drains constructed in the landfill itself. These drains would consist of perforated drain pipe surrounded with gravel. These collector pipes would be inclined toward a central sump equipped with a small pump to remove an anticipated 45,000 gpd to a leachate treatment facility.

It is generally agreed that the most promising leachate collection system would consist of several recovery wells located within, directly beneath, or in immediate proximity of the landfill so as to maximize the collection of leachate and minimize the collection of clean water. This conclusion is tempered to the need for pump tests in the landfill area to determine local well interference and subsequently determine the location and number of recovery wells necessary to insure virtually complete leachate recovery.

In considering this alternative, one must address the disposal of recovered concentrated leachate. It has been suggested that the recovered leachate could be discharged into a County sewer located in the immediate proximity of the landfill. Initial capital costs associated with this alternative are approximately $4,450,000 with operation and maintenance costs reaching $990,000 per year.

III. REMOVE THE SOURCE

Several alternatives have been considered with respect to removing the source of contamination, including the following:

A. Transport to Another Landfill - This alternative consists of excavating the entire landfill and transporting it to another site with adequate environmental safeguards.

The major shortcoming of this alternative is that much the same hydrogeologic control and leachate treatment would be required at the new site as would be required at the present site. Additionally, there are environmental unknowns and significant costs associated with excavating the landfill. Initial capital costs for the transporting alternative are approximately $14,770,000. Yearly operation and maintenance costs would be approximately $970,000.

B. Incinerate - This alternative consists of excavation, incineration, and removal of the residue to another site. Of the incinerators investigated, the Union Carbide Purox System (with or without shredding) and a steam generating incinerator have been retained as viable alternatives as long as fresh refuse to reduce overall moisture content is available. Among the considerations to be made about this alternative are excessive capital costs requiring integration of the incineration facility into a regional solid waste management program, the need for a supplemental fuel source, the need for air pollution control measures and operational difficulties. Initial capital outlay would range from $22.7 to $38.3 million. Yearly operation and maintenance costs would range from $1.17 to $1.8 million.

IV. HASTEN DECOMPOSITION

Several alternatives were considered to hasten in situ decomposition of the refuse.

A. Recirculation - Spray irrigation of treated domestic sewage and industrial wastewaters has been demonstrated to be an effective means of waste treatment. The treatment processes include aeration while spraying, bacterial decomposition in the upper soil layers, chemical adsorption on soil particles, and chemical uptake by plants.

Unfortunately, spray irrigation of leachate would have several major difficulties, including:

1. Airborne odors.
2. Possible chemical

toxicity to plants.
3. Clogging of surface soil pores and plant stomata by iron and manganese precipitates from the oxidized leachate.
4. Clogging of surface soil pores by a slime derived from the highly organic chemical loading in the leachate.

The highly concentrated nature of the leachate would require that it be sprayed onto the landfill at rates considerably lower than the 2-inch per week guideline for treated domestic wastewater on permeable soils. If the leachate were recycled by spreading through infiltration galleries on the landfill surface, clogging of these galleries by aerated leachate precipitates would still be a problem.

Addition of water to the landfill would generate large volumes of contaminated water; some of which would inevitably leak through the landfill floor to the Potomac aquifer. The addition of water would also raise the zone of saturation in the landfill, causing outflow of leachate to the Columbia sand north of the landfill by local reversal of the water table gradient, and into Army Creek as it seeps along the western, southern, and perhaps eastern margins of the landfill.

Thus, with no assurance that recycling water into the landfill would appreciably hasten the achievement of chemical stability of the buried refuse and considering the problems discussed above, the most troublesome of which is the production of large volumes of leachate, such a program should not be initiated.

B. Annelidic Consumption - This alternative involves the use of earthworms to metabilize organics present in the landfill. Since earthworms can only survive in an aerobic environment, excavation will be an integral step in this process. Metabolized materials would be removed in a stepwise fashion after the earthworms have left a section of the fill. It is also realized that the costs associated with earthworms and earthworm management addresses only a small fraction of the refuse and the vast majority of the refuse must be later disposed of off-site. Initial capital costs would be \$19.8 million and operation and maintenance costs would approach \$870,000 per year.

V. PRESSURE MAINTENANCE

Rather than recovering contaminated water from beneath the landfill, another alternative was to inject fresh water beneath the landfill to raise the peizometric surface of the Potomac aquifer to the landfill base, thus preventing the downward migration of contaminants into the Potomac Formation. Water could be drawn from elsewhere in the aquifer for injection at the landfill site. This alternative would require a rather extensive pumping/injection well network with associated operational problems. It would be necessary to recover leachate within the landfill if this correctional measure was undertaken.

VI. LANDFILL AERATION

This alternative involves the bubbling of air into the landfill. There are indications that this process, coupled with minimization of leachate production, is effective in precipitating heavy metals and enhancing aerobic stabilization of other contaminants. The precipitated metals could be of assistance in sealing the base of the landfill, thereby reducing the downward migration of contaminants. Concern has been expressed over how well air could be distributed through the refuse. It was also anticipated that operational difficulties would be encountered.

RECOMMENDED ACTION

In November, 1977, a panel of nationally recognized authorities in the field of Solid Waste Disposal reviewed the alternatives developed by New Castle County. It was the consensus of these participants that the following measures be instituted to effectively manage the problem.

1. Minimize Leachate Production

a. Minimize precipitation infiltration through the refuse by the application of a relatively impermeable cover material, regrading the surface to promote stormwater runoff, and revegetation.

b. Minimize ground water inflow through the refuse by excavating a cutoff trench along the northwestern boundary of the landfill.

2. Maximize Leachate Recovery

a. Construct new recovery wells within or closer to the landfill boundary to recover concentrated leachate before it enters the Upper Potomac Formation.

b. Phase out existing recovery well system as downgradient water quality improves. This action will ultimately make available additional water for production purposes.

To this end, New Castle County and their consultant, R. F. Weston, Inc., have developed a five year implementation plan for which we are presently seeking funds. It is hoped that Federal and State participation will permit the completion of this project by 1984.

References: LEIS, W. M. and D. C. CLARK. Control Program for Leachate Affecting a Multiple Aquifer System, Army Creek Landfill, New Castle County, Delaware 1976.

Army Creek Landfill Technical Roundtable, November 18, 1978, Summary Proceedings, New Castle County Areawide Waste Treatment Management Program. Unpublished Draft.

PREDICTING MOVEMENT OF SELECTED METALS IN SOILS: APPLICATION TO DISPOSAL PROBLEMS

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ABSTRACT

Predicting movement of polluting constituents requires the establishment of a model whether it is "mental", based wholly on experience, or mathematical, based on sophisticated computerizable equations derived from actual migration data. The Lapidus and Amundson mathematical model and its solution as proposed was adopted in the prediction of movement of Cd, Ni, and Zn using parameters from disturbed soil columns and municipal solid waste leachates to develop a user-oriented predictive tool. The cadmium prototype is presented in this paper to illustrate its use. Confidence in this approach for application to disposal requires verification under field conditions.

BACKGROUND

The ability to predict movement of pollutants through soil must become one of our main goals in disposal operations, since the soil is considered as the ultimate receptacle of most of our nation's waste. Functional predictions are not beyond reality. The soil scientist, for example, well demonstrates the potential for such predictions in the field of plant nutrition, soil reclamation and salt control (Allaway, 1968; Lisk, 1972; Walsh and Beaton, 1973, USDA, 1954). Unfortunately, direct application of these studies to migration rates of trace and heavy metals in waste streams, particularly the ones involving the conventional cation exchange capacity and mild extractions (Black, 1965), has not been found to share the same success (Korte et al., 1976). In fact, the hydrous oxides of Fe, Mn, and Al appear to be as influential, if not more so, to retention of heavy and trace metals in soils, as cation exchange capacity (Jenne, 1968; Korte et al., 1975, 1976; and Leeper, 1978). Furthermore, waste streams involved a broader spectrum of elements than plant nutrients, and usually are carried in a wide diversity of solutions which affects attenuation in a dominant manner.

Before predictive designs for disposal of toxic-metal-containing wastes can be established, a multitude of factors influencing metal migration rates in soils must be identified and evaluated. The three prominent components of the disposal environment affecting metal migration are soil, leachate, and specific element (Fuller, 1977). No two metals respond (attenuate) identically to the same set of parameters.

Prediction of pollutant migration rate through soils may be approached in several different ways. The most common method available to us at this time depends strictly on personal judgment. In the selection of a suitable site for disposal, the operator must apply a mental "model" for the evaluation of the many factors he keeps in mind as a result of experience. A mental "model" is the same as a theoretical and computerized model minus the hardware and is subject to errors of judgment based on qualitative rather than quantitative information. Thus, a model is a simplified representation of an actual waste disposal system (van Genuchten, 1978) whether it exists in the mind from practical experience or is computerized using measured parameters and complicated equations.

Another means of predicting pollutant migration rate is to rank the pollutants according to soil interactions as illustrated by Korte et al. (1976). Although this method is more quantitative than the mental "model", it too is a highly qualitative procedure. Heavy metal-soil interactions were ranked (Figures 1 and 2) using 11 prominent toxic waste elements, As, Be, Cd, Cr, Cu, Hg, Ni, Pb, Se, V, and Zn with 10 soils representing 7 of the 10 major soil orders of the world (Table 1). Cations and anions were ranked separately because of differences in migration behavior in different soils. Changes in ordering of soils when going from cations to anions involve a higher rank for soils having lower pH values and/or higher free iron oxide content. The soil environment was strictly anaerobic and the leachate flow saturated, like that of conditions under a landfill operation. Predicted relationships for similar soils and any situation can be found by using Tables 1 and 2 and typical attenuation curves as presented for each soil and a given element (Korte et al., 1976), Figures 3 and 4.

The "models" in the above cases are highly empirical and seriously lacking in the precision necessary for long-range predictions in pollution control. Therefore, we have selected a mathematical model patterned after that of Lapidus and Amundson (1952) in simulating metal migration through soil. Soil and broad leachate properties that most influence metal attenuation have been quantitatively measured as previously described in detail by O'Donnell et al., 1977; Korte et al., 1976; and Fuller, 1977, 1978. These measurements are in the form of stochastic functions of

TABLE 1. CHARACTERISTICS OF THE SOILS USED.

Soil series*	Soil texture Clay	 Silt	 Sand	Texture class USDA	pH soil paste	Cation exchange capacity	Elec. Cond. of extract	Column bulk density	Porosity	Total surface area
	%	%	%			meq/100g	µmhos/cm	g/cm^3		m^2
Davidson	61	20	19	clay	6.2	9	169	1.89	0.476	18,212
Molokai	52	25	23	clay	6.2	14	1262	1.44	0.429	23,420
Nicholson	49	47	3	silty clay	6.7	37	176	1.53	0.460	43,019
Fanno	46	19	35	clay	7.0	33	392	1.48	0.484	44,811
Mohave (Ca)	40	28	32	clay loam	7.8	12	510	1.54	0.446	50,235
Ava	31	60	10	silty clay loam	4.5	19	157	1.45	0.478	22,325
Anthony	15	14	71	sandy loam	7.8	6	328	2.07	0.360	8,831
Mohave	11	37	52	sandy loam	7.3	10	615	1.78	0.365	16,507
Kalkaska	5	4	91	sand	4.7	10	237	1.53	0.404	3,542
Wagram	4	8	88	loamy sand	4.2	2	225	1.89	0.378	3,616

*Oriented on basis of clay content.

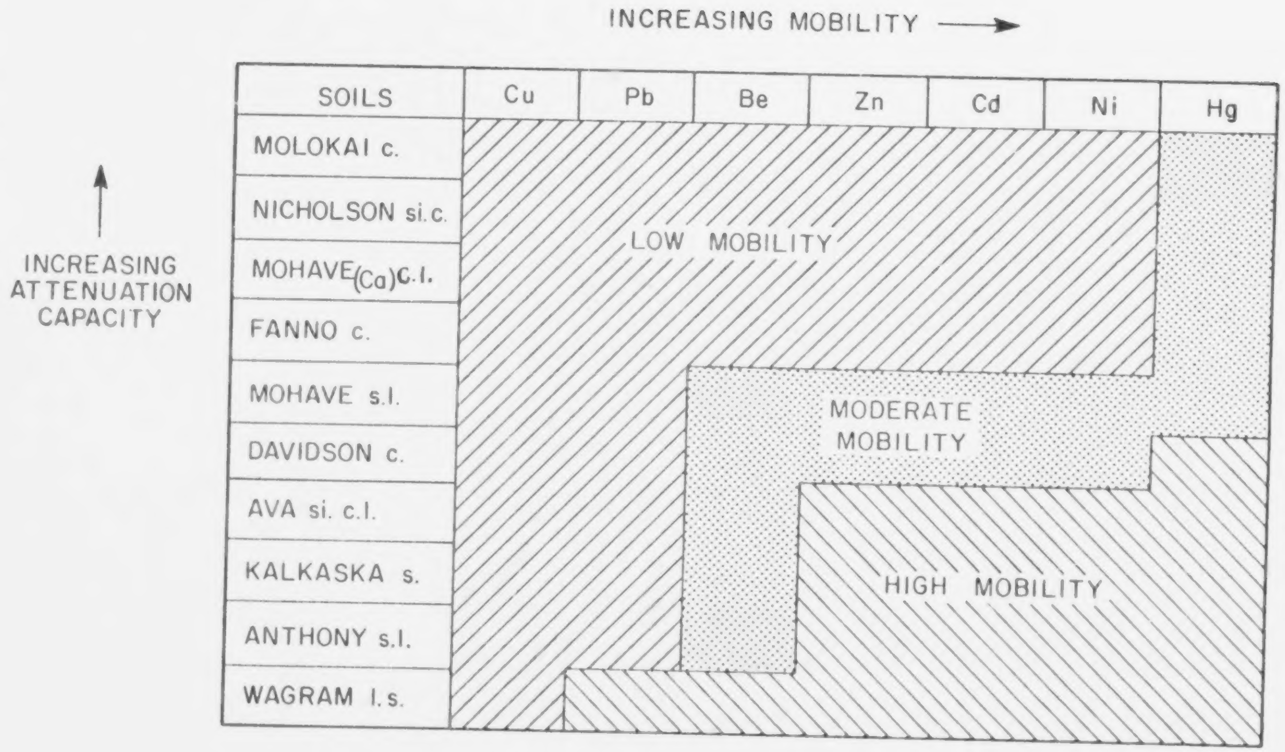
INCREASING MOBILITY →
INCREASING ATTENUATION CAPACITY ↑
SOILS
Cu
Pb
Be
Zn
Cd
Ni
Hg
MOLOKAI c.
NICHOLSON si.c.
MOHAVE(Ca)c.l.
FANNO c.
MOHAVE s.l.
DAVIDSON c.
AVA si. c.l.
KALKASKA s.
ANTHONY s.l.
WAGRAM l.s.
LOW MOBILITY
MODERATE MOBILITY
HIGH MOBILITY

Figure 1. Relative mobility of cation-forming elements through soil.

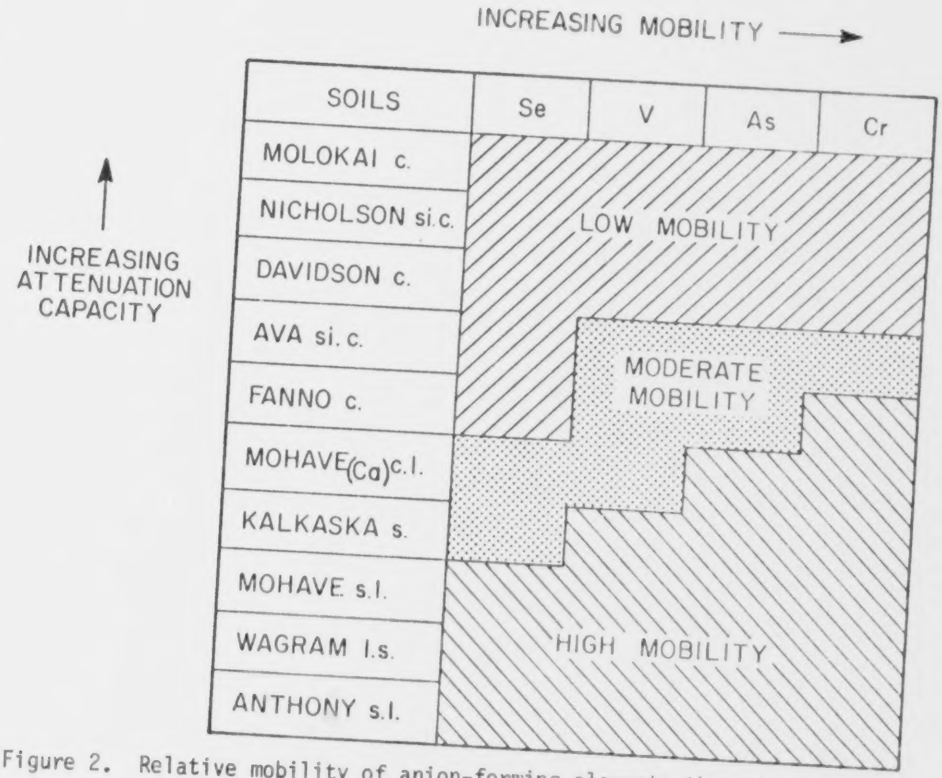
INCREASING MOBILITY
INCREASING ATTENUATION CAPACITY
SOILS
Se
V
As
Cr
MOLOKAI c.
NICHOLSON si.c.
DAVIDSON c.
AVA si.c.
FANNO c.
MOHAVE(Ca) c.l.
KALKASKA s.
MOHAVE s.l.
WAGRAM l.s.
ANTHONY s.l.
LOW MOBILITY
MODERATE MOBILITY
HIGH MOBILITY

Figure 2. Relative mobility of anion-forming elements through soil.

TABLE 2. DESIGNATION SHOWING TYPE OF CURVE GENERATED FROM EACH COLUMN

Soil	As	Be	Cd	Cr	Cu	Hg	Ni	Pb	Se	V	Zn
Wagram	A*	C	A	A	E	A	A	B	D	A	A
Ava	C	D	A	C	D	B-C	A	D	E	B	A-B
Kalkaska	C	D	C	C	E	C	B	E	C	D	B
Davidson	C	D	C	E	E	C	B	E	E	E	D
Molokai	E	E	E	E	E	D	D	E	E	E	E
Chalmers	-†	-	-	-	-	D	D-E	-	-	C	-
Nicholson	E	E	E	D	E	B-C	E	E	E	E	D
Fanno	C	E	E	C	E	C	E	E	E	D	C
Mohave	B	D	D	A	E	B	C	E	C	B	B
Mohave (Ca)	E	E	E	A	E	D	E	E	C	E	E
Anthony	A	D	A	A	D	A	A	D	C	A	B

*A, B, C, ... refer to curves in Figure 4.

†- designates experiments not run.

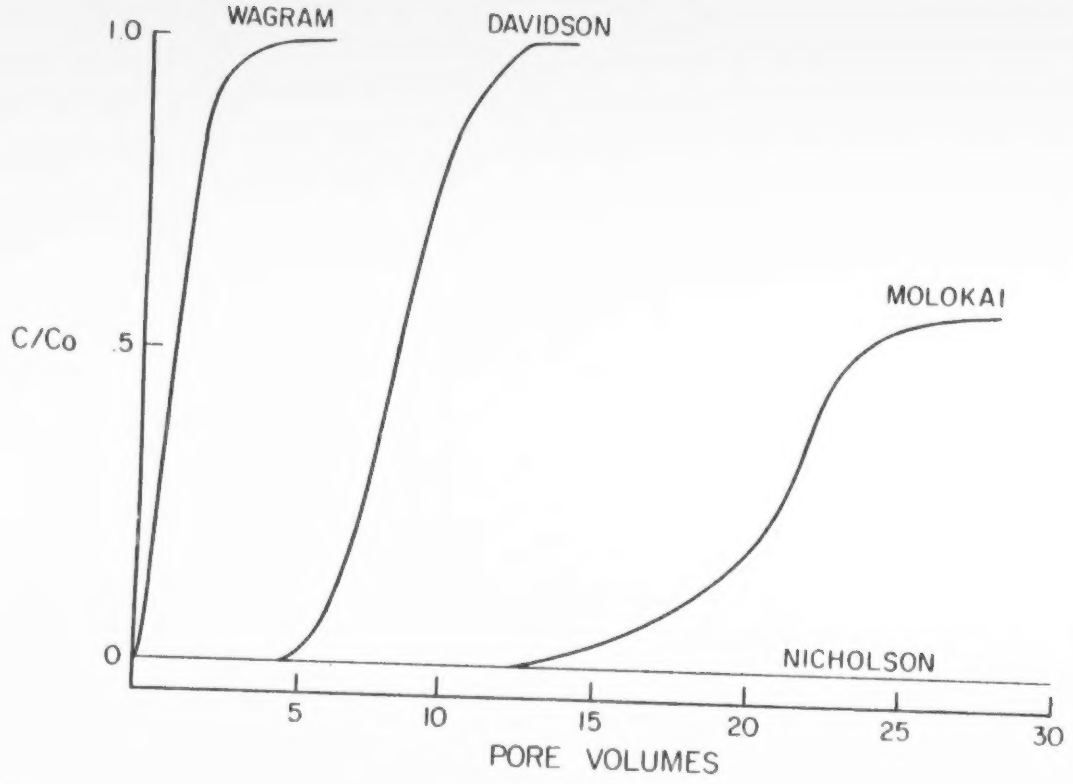

Figure 3. Relative migration of Ni through four soils.

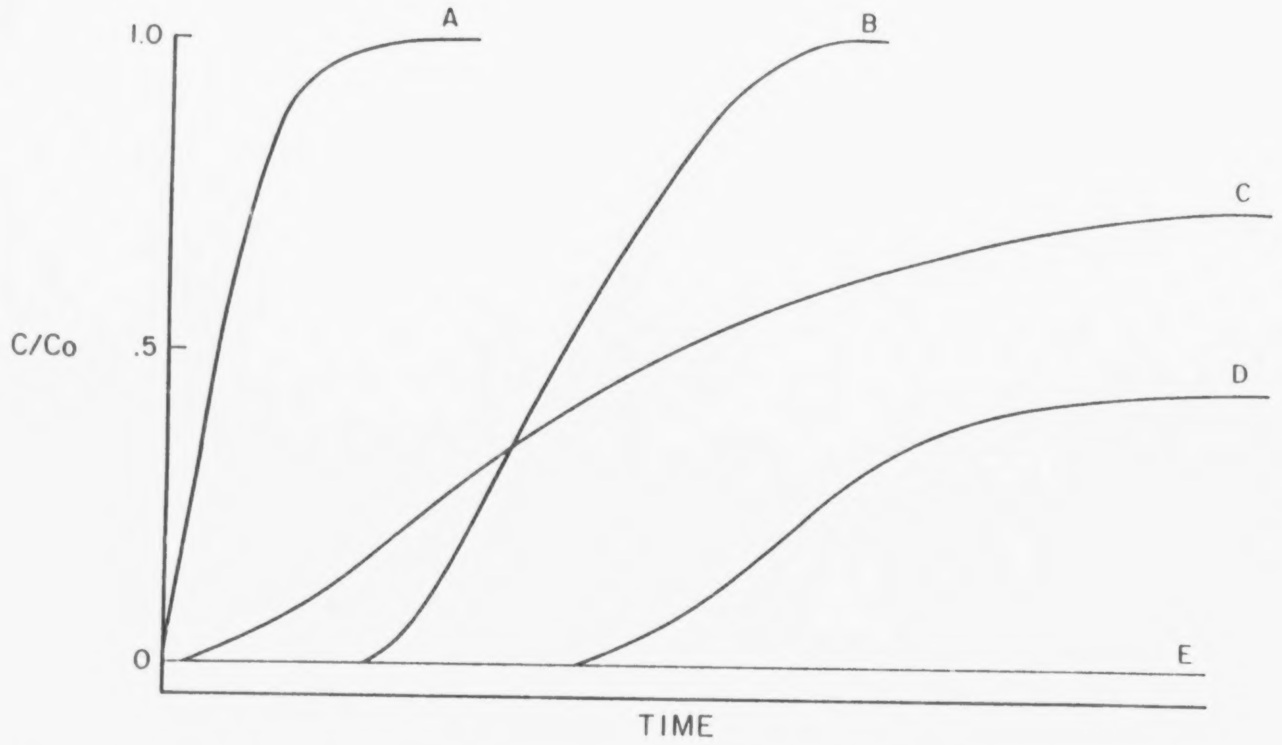

Figure 4. Types of breakthrough curves generated (horizontal scale is approximately 30 days).

specific properties of soil-leachate-metal systems. The general plan has been to match theoretical curves describing selected metal movement in soil columns under saturated flow conditions with actual data. Forcing such a match gives rise to specific values for the theoretical parameters in the L-A model. For each soil column, these parameters can be translated directly into a migration rate for a particular element (metal).

OBJECTIVE

The major objective is to determine the long-range, steady-state migration rates of selected metals contained in landfill-type leachates and to arrive at simple formulas that will predict migration rates.

APPROACH

To examine metal migration rates under field conditions is difficult and very time consuming. Useful data for screening purposes may be obtained through the employment of soil columns in the laboratory (Fuller, 1977, 1978), while avoiding the problems of field conditions. Results of this research may then be exposed to field experiment for further verification and necessary refinements. We have elected to use soil column studies in connection with a mathematical model (O'Donnell et al., 1977).

Many models have been proposed to describe chemical movement in porous media. Selim and Mansell (1976) considered solute flow in finite columns where the solute and adsorbate followed an instantaneous equilibrium relationship. Skopp and Warrick (1974) considered the soil to be composed of a mobile and stagnant region where the solute adsorption is a diffusion controlled process (Figure 5). Van Genuchten and Wierenga (1976) considered the soil to be composed of several regions. The solute adsorption, however, was considered to be a function of the concentration gradient between the mobile and stagnant regions. Models by Selim et al. (1976), and Davidson et al. (1975), had numerical solutions.

Any of the above models can be used to describe metal transport in the soil. All have some unknown parameters which must be evaluated from experiemntal data. This involves tedious computer work if the solutions are complicated. The model and its solution proposed by Lapidus and Amundson (1952) has an advantage that it is simple (relative to Selim and Mansell, 1976, and van Genuchten and Wierenga, 1976) and it has an analytical representation in contrast to Davidson et al. (1975), and Selim et al. (1976).

The Lapidus-Amundson mathematical model is:

$$\frac{\partial c}{\partial t}+\frac{1}{\alpha}\frac{\partial n}{\partial t} = D\,\frac{\partial^2 c}{\partial z^2} - v\,\frac{\partial c}{\partial z} \qquad (1)$$

where:

c = concentration in soil solution (M/L^3)

v = convective (pore water) velocity (L/T)

D = diffusion coefficient (L^2/T)

α = fractional pore volume of soil (L^3/L^3)

n = amount adsorbed per unit volume of soil (M/L^3)

z = vertical distance (L)

t = time (T)

The term $\partial n/\partial t$ describes a linear, non-equilibrium adsorption and is assumed to be

$$\partial n/\partial t = K_1 c - K_2 n \qquad (2)$$

where K_1 and K_2 are forward and backward reaction rates ($1/T$), respectively.

The boundary and initial conditions for a steady input concentration (c_0) are:

$$c = c_0 \quad , \quad z = 0 \quad , \quad t > 0 \qquad (3a)$$

$$\left.\begin{matrix} c = 0 \\ n = 0 \end{matrix}\right\} \quad , \quad t = 0 \quad , \quad z > 0 \qquad (3b)$$

The solution to Eqs. (1) and (2) subject to (3a) and (3b) is

$$c = c_0 \exp(vz/2D)[F(t) + K_2 \int_0^t F(t)dt] \qquad (4)$$

where

$$F(t) = [\exp(-K_2 t)] \int_0^t \{I_0[2/\alpha)\sqrt{K_1 K_2 \beta(t-\beta)}]$$
$$\exp[-(z^2/4D\beta + \beta d)]\cdot[z/2\sqrt{\pi D\beta}]\}d\beta$$

with $d = v^2/4D + K_1/\alpha - K_2$. I_0 is the modified Bessel function of the first kind of order zero. The term c/c_0 is referred to as

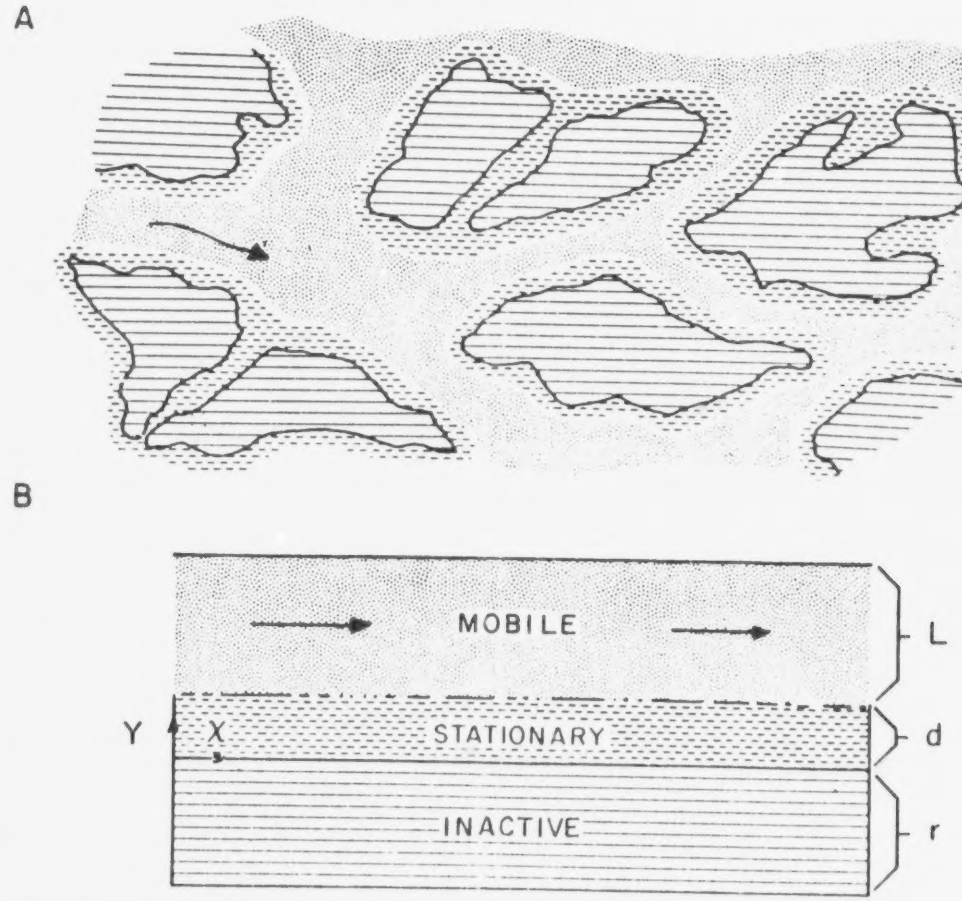
A
B
MOBILE
STATIONARY
INACTIVE
L
d
r
Y
X

Figure 5. Idealized flow regime in porous media. (A) Model of flow in soils, (B) Simplified model, analogous to A. The shading patterns in A and B correspond.

relative concentration, C, and is a function of z and t, and contains the parameter v, α, D, K_1 and K_2.

The Lapidus-Amundson model was formulated to describe the effect of longitudinal diffusion in ion exchange and chromatographic columns, where the chemical and physical parameters are uniform and well controlled; and the system is relatively simple. In a soil-leachate system, the number of parameters increases greatly, and their interrelationships are much more complex. The advantage of the Lapidus-Amundson model is that it combines all unknown or immeasurable parameters into three unknowns, D, K_1, K_2, which can be determined from the experimental data as described below. It would be impossible to determine the spatial interrelation between all the different physical and chemical sites in the soil-leachate system. By applying the Lapidus-Amundson model, the net effect of the chemical nature of the leachate on the forward and backward reaction rates at a multitude of different adsorption sites is combined into an effective forward and backward reaction rate for the particular conditions of the soil-leachate system in the experiment. Similarly, an effective diffusion coefficient is determined, or a diffusion-dispersion coefficient since the effects of diffusion-dispersion combine in one variable.

Miscible displacement theories and experimental data indicate the solute profile is not one of piston displacement, but rather is a smooth distribution of concentrations (Nielsen and Biggar, 1962). In addition, different relative concentrations appear to travel at different rates (Figure 6). Since the velocity of a particular concentration ratio is a function of distance, it follows that the shape of the experimental breakthrough curve will also be a function of z. However, the model predicts that for $z > 10$ cm the velocity of a particular relative concentration approaches an asymptotic value; i.e., steady-state velocity for any relative concentration has been achieved by a depth of 10 cm (Figure 6).

Six U.S. soils representing five of ten major soil orders were used to make 10-cm soil columns which were leached with two municipal solid waste leachates (Fuller, 1978) whose salt and iron contents had been altered to varying degrees (Table 3).

In the application of the model, v is estimated by v_d/α where v_d is the Darcian velocity. The variables α, c, c_0 are easily measured. Using these measurable parameters the values of D, K_1, K_2 are selected to give the closest match between the breakthrough curve predicted by the model and the experimental breakthrough observed in the column experiments. A variable metric minimization routine (Davidon, 1959) was employed to find the location of the minimum of the sum of squares of the differences between the experimental values, $Y(t_i)$ and the predicted values $C(t_i)$; i.e.,

$$SSD = \sum_{i=1}^{n} [Y(t_i) - C(t_i)]^2$$

To obtain the theoretical breakthrough curve, the Eq. (4) is integrated numerically using a 16-term Gaussian Legendre quadrature formula (Abramowitz and Stegun, 1970). The modified Bessel function, I_0, is approximated by a Chebyshev polynomial (Abromowitz and Stegun, 1970). Fifteen equally spaced points from the experimentally determined breakthrough curve were used for comparison.

The program used is iterative and may be continued until the sum of squares of errors (SSD) is as small as desired. When this point is reached, the current values of D, K_1 and K_2 are accepted as the true values.

Once the parameters D, K_1 and K_2 for a specific soil and a specific element are determined, the relative concentration (i.e., $c/c_0 = C$) is a function of z and t only. Our attempt is, however, to determine the migration rate of a particular relative concentration. The question is, knowing all the parameters (D, K_1, K_2, α, v and c_0) for a given soil and a given element, how long will it take for a particular relative concentration, C, to reach a given depth, say ground water. Mathematically, what we are asking is the following: Find a function $z = z(t)$ so that for any time, $t > 0$, the following is true:

$$C(z(t), t) = C'$$

Thus, for any time $t > 0$, $z(t)$ gives the location of the relative concentration C'. The migration rate of C' would then be

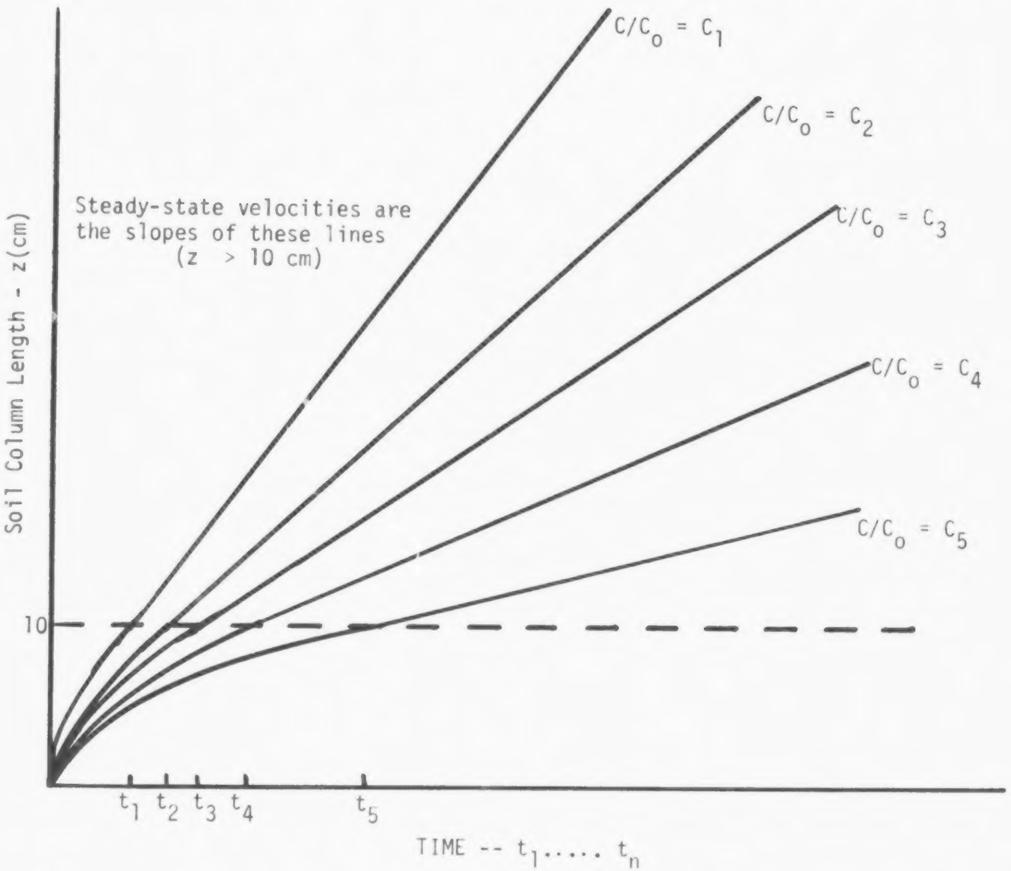

Figure 6. Trajectories of different relative concentrations. (Plot of $z = z(t)$, where $z = z(t)$ satisfies $c(t,z(t)) = C_i$, $i = 1,2,3,4,5$.

TABLE 3. SOME CHEMICAL CHARACTERISTICS OF LEACHATES REMOVED FROM A MUNICIPAL SOLID WASTE LANDFILL GENERATOR, TUCSON, AZ: LANDFILL II.

Leachate* sample date	pH	EC	TOC	Ca	Mg	Na	K	Fe	Mn	Zn	Co	Ni	Si	PO_4
		µmhos/cm	ppm											
8/29/76	5.4	17,000	11,628	820	310	418	1060	885	12.5	6.5	0.6	0.70	33	24
9/27/76	5.4	11,500	10,082	815	270	374	980	900	12.2	8.0	0.6	0.70	36	31
10/20/76	5.4	9,000	7,628	690	100	300	700	640	12.0	6.0	0.5	0.50	33	24
11/8/76	5.4	9,000	7,491	731	107	295	700	720	12.3	6.8	0.5	0.25	26	24
12/1/76	5.4	9,500	7,446	760	108	305	695	720	12.5	7.8	0.4	0.25	25	24
1/14/77	5.4	9,500	7,765	760	108	295	680	720	13.0	9.3	0.4	0.25	25	24
2/8/77	5.4	9,100	7,949	665	105	295	670	785	13.0	9.3	0.4	0.25	25	24

*Dates during which soil-column experiments were undertaken: Cd (8/29/76-9/2[illegible]/76). Cu, Cd, Cr, and Pb were below detectable limits.

TABLE 4. APPARENT Cd VELOCITY AS RELATED TO PORE WATER VELOCITY. (Calculated from equation (10a) with parameters $z = 10$ cm, $\alpha = .35$, $D = 2\ cm^2/day$, $K_1 = 7\ day^{-1}$, $K_2 = 15\ day^{-1}$.)

Pore Water Velocity	C/C_o									
	0.1	0.2	0.3	0.4	0.5	0.6	0.7	0.8	0.9	0.95
3 cm/day	1.65	1.55	1.48	1.42	1.36	1.30	1.24	1.14	0.89	--
6 cm/day	3.07	2.90	2.80	2.70	2.62	2.53	2.43	2.28	2.01	--
12 cm/day	6.02	5.57	5.50	5.37	5.17	5.00	4.73	4.54	4.34	4.16
24 cm/day	12.83	11.94	10.84	10.70	11.10	9.74	8.55	8.21	8.20	7.62

given by dz/dt.

The function $C(z(t), t) = C'$ is complicated, therefore it is not possible to solve for $z(t)$ directly. However, if one fixes not only the relative concentration at C' but also chooses a particular depth z, it is possible to use an interpolation scheme to find the time t at which the relative concentration C' passes the depth z. That is, this interpolation scheme determines one point along the curve $z = z(t)$. By choosing various fixed depths and using the same interpolation methods, one obtains a series of points along the curve $z = z(t)$, Figure 6. Note that in general $z = z(t)$ is not a straight line, however, for depths greater than about 10 cm, the curve becomes linear for all practical purposes. That is, it achieves a steady-state condition.

By estimating the steady-state velocity using the "linear" portion of the curve, we can determine the effect of change in the soil-leachate system on migration rates using multiple regression analysis. The main characteristics of the soils and leachates have been measured to identify the effect of soil and leachate properties on Cd, Ni, and Zn attenuation. The soil parameters are, a) texture, b) pH, c) free iron, d) total dissolved solids or salt (ION), and e) surface area; for leachates they are, a) total organic carbon (TOC), b) inorganic salts (ION), c) soluble Fe, d) pH.

Using Cd movement as prototype, Table 4 lists the calculated speed of the relative concentrations for the parameters $z = 10$ cm, $\alpha = 0.35$, $D = 2$ cm^2/day, $K_1 = 7$ day^{-1} and $K_2 = 15$ day^{-1}. The table entries are Cd velocities as they move past $z = 10$ cm for leachate pore velocities of 3 cm/day, 6 cm/day, 12 cm/day and 24 cm/day. Most of the velocities follow the simple rule of being halved when the leachate velocity is halved. So we assume that the velocity of a given concentration ratio is proportional to the pore velocity of the leachate and can therefore put the Cd velocity at a particular relative concentration C' and particular fluid speed, s, as:

$$\begin{aligned} \mathrm{Vel}(C',s) = S(&\mathrm{A1}\cdot\mathrm{FACTR\text{-}1} + \mathrm{A2}\cdot\mathrm{FACTR\text{-}2} + \ldots \\ &+ \mathrm{AN}\cdot\mathrm{FACTR\text{-}N}) \end{aligned} \qquad (5)$$

The factors FACTR-i, $i = 1, \ldots, N$, in Eq. (5) are the particular soil leachate properties under consideration.

The application of this type of equation is of course only valid if the migration rate is proportional to the pore velocity of the carrier fluid. Otherwise a multitude of additional experiments will need to be undertaken with pore velocity as an additional variable. But preliminary experiments indicate that over at least a limited range, this proportionality is maintained.

It was determined in the course of our experiments that it is the ratio of the forward and backward reaction rates which is the dominant factor in determining the breakthrough curve, as opposed to the absolute magnitude of K_1 or K_2. A plot of the values of K_1 and K_2 which give an acceptable fit of the theoretical breakthrough curve to the experimental curve results in a straight line with a slope equal to K_1/K_2. The line segments generated for each of the soils reported varied within a narrow range if at all at different fluid velocities. For example, when flux was changed fourfold (i.e., 3 ml/hr to 12 ml/hr) the K_1/K_2 ratios varied according to a common trend but to a limited extent for the soils involved (see Figure 7). Such narrow ranges of differences in slope of line segments as a result of differences in fluid velocities are not universal for all soils. Nicholson sic and Molokai c are notable exceptions according to soils from our laboratory not reported here. Nevertheless, for the purpose of model development we chose to assume flux is of much less importance as a variable than other parameters in the soils used in this research program (Alesii et al. 1979).

The flux dependency of the diffusion coefficient also needs to be considered. A true diffusion coefficient should be independent of pore velocity (Bresler, 1973). But since the D term in the Lapidus-Amundson equation is actually a combination diffusion-dispersion term, there will be some flux dependence since the dispersion coefficient is a function of pore velocity. The diffusion coefficient has been shown to be independent of the salt concentration and dependent on only the water concentration for all practical purposes in soil-water systems (Porter et al., 1960; Kemper and van Schaik, 1966). Kemper and van Schaik (1966) propose a functional relationship between the diffusion coefficient, D, and the volumetric water content, θ, as being $Dp(\theta) = D_0 a e^{b\theta}$; where a and b are

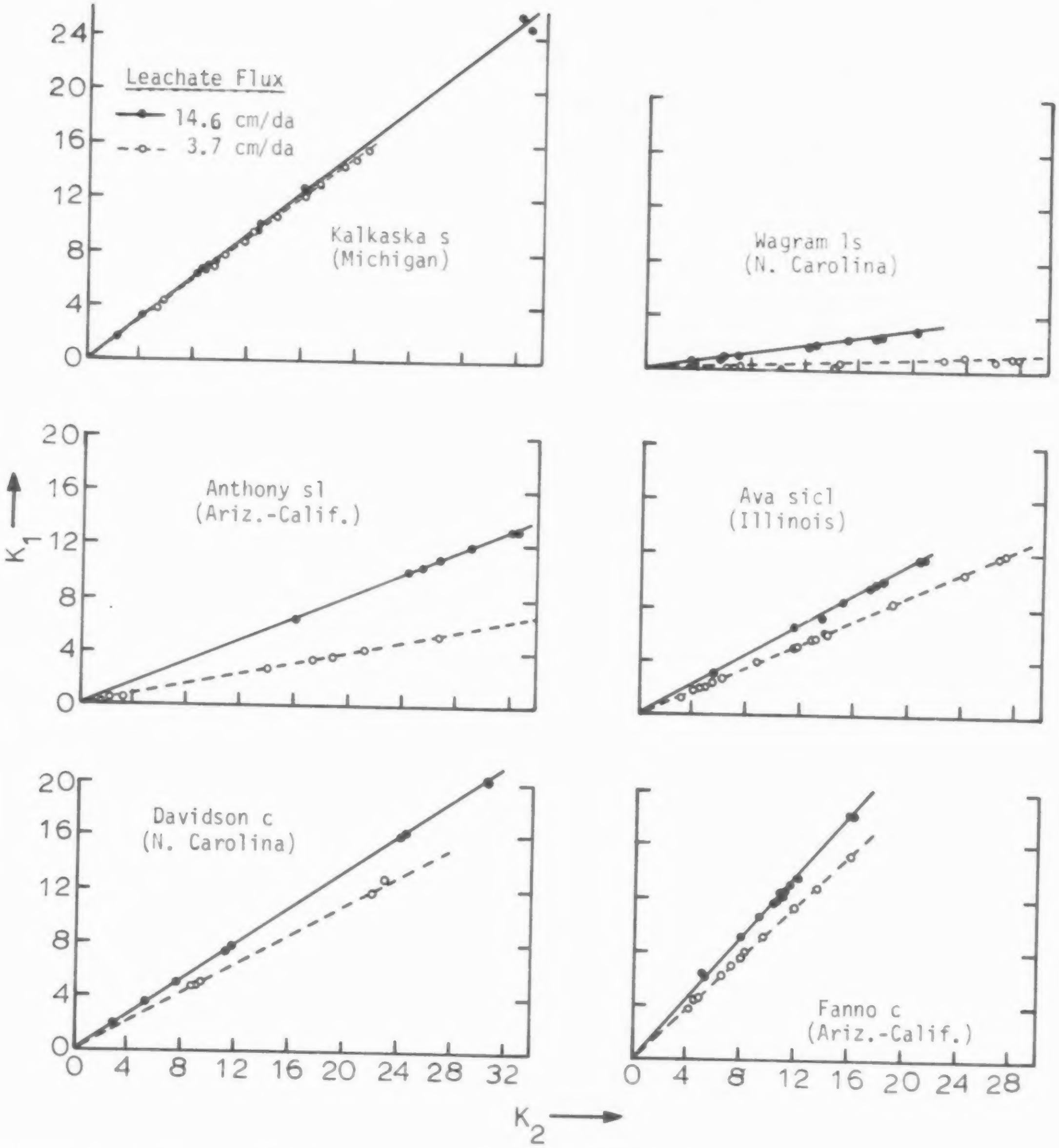

Figure 7. Line segments for six soils relating forward (K_1) and backward (K_2) reaction coefficients as influenced by flux, where: Rate of Adsorption $\partial n/\partial t = K_1c - K_2n$ (Eq. 2).

empirical constants, and D_o is the diffusion coefficient in a free-water system. This relationship has been found to hold when the range of water content corresponds to 0.30 to 15 bars suction by Olsen and Kemper (1968), when b equals 10 and a ranges from 0.001 in clays to 0.005 in sandy loams. These values for a and b were also successfully applied by Bresler (1973), and Melamed et al. (1977).

Bresler (1973) reports that many investigators have shown a linear relationship between the dispersion coefficient D_h and the average pore velocity v in the form $D_h(V) = \lambda|v|$ where λ is an empirically determined constant. Bresler (1973) reports values for λ of 0.28, 0.39, and 0.55; while Melamed et al. (1977) use a value of 0.40.

The net result is that the diffusion - dispersion term, D, is related to D_p and D_h as follows :

$$D(\theta,V) = D_p(\theta) + D_h(V) \tag{6a}$$

$$D(\theta,V) = D_o a e^{b\theta} + \lambda|V| \tag{6b}$$

In the experiments which we did, the values for v were on the order of 0.82 cm/hr in Davidson clay and 1.17 cm/hr in Wagram loamy sand. Using these flux rates and the D_o of chloride (0.04 cm^2/hr) as a reference point, it can be seen from the relation above that D_h is more than an order of magnitude greater than D_p (0.23 cm^2/hr vs 0.0048 cm^2/hr in Davidson clay). In the flux range in which the experiments were run, the equation predicts that D_h describes 98% of D. By decreasing the flux by an order of magnitude, the D_h term should still describe 83% of D. We assume therefore for the purposes of model development that the relationship between the diffusion-dispersion coefficient and flux α remains linear over the range of interest. Biggar and Nielsen (1976) present a relation between the apparent diffusion coefficient, D, and pore water velocity v, $D = 0.6 + 2.93\ v^{1.11}$ with an r value of 0.795.

We regress the soil and leachate properties having most effect on migration rates with the migration rates in the following way: The regression equation is written one variable at a time with all variables available to bring into the equation at each step. The first variable in the equation is that one which does the most to explain the variation in the migration rates. The second variable to enter the equation is that one which does the most to explain the variation in the migration rates not already explained by the first variable. Variables continue to be added in this way until either all variables are in the equation or until none of the unused variables are statistically significant in explaining the remaining variation in the migration rates.

INTERPRETATION OF RESULTS

The apparent cadmium velocity through the soil is approximated by the following set of equations:

$$V\ .1 = \frac{V}{30}[-1.589(\text{CLAY}) + 0.0196(\text{CLAYxCLAY}) + 1.385\ (\text{IONxSILT}) - 14.054(\text{FeO}) + 0.207(\text{CLAYxFeO}) + 0.0122(\text{CLAYxSILT}) + 37.667]$$

$$V\ .2 = \frac{V}{30}[-1.472(\text{CLAY}) + 0.018(\text{CLAYxCLAY}) + 1.238(\text{IONxSILT}) - 12.504(\text{FeO}) + .183(\text{CLAYxFeO}) + 0.011(\text{CLAYxSILT}) + 34.022]$$

$$V\ .3 = \frac{V}{30}[-1.401(\text{CLAY}) + 0.018(\text{CLAYxCLAY}) + 1.187(\text{IONxSILT}) - 11.511(\text{FeO}) + 0.168(\text{CLAYxFeO}) + 0.011(\text{CLAYxSILT}) + 31.693]$$

$$V\ .4 = \frac{V}{30}[-1.347(\text{CLAY}) + 0.017(\text{CLAYxCLAY}) + 1.166(\text{IONxSILT}) - 10.809(\text{FeO}) + 0.157(\text{CLAYxFeO}) + 0.010(\text{CLAYxSILT} + 29.924]$$

$$V\ .5 = \frac{V}{30}[-1.289(\text{CLAY}) + 0.016(\text{CLAYxCLAY}) + 1.121(\text{IONxSILT}) - 10.245(\text{FeO}) + 0.149(\text{CLAYxFeO}) + 0.010(\text{CLAYxSILT}) + 28.391]$$

$$V\ .6 = \frac{V}{30}[-1.190(\text{CLAY}) + 0.015(\text{CLAYxCLAY}) + 1.069(\text{IONxSILT}) - 10.190(\text{FeO}) + 0.149(\text{CLAYxFeO}) + 0.010(\text{CLAYxSILT}) + 27.137]$$

$$V\ .7 = \frac{V}{30}[-1.184(\text{CLAY}) + 0.015(\text{CLAYxCLAY}) + 0.986(\text{IONxSILT}) - 9.272(\text{FeO}) + 0.134(\text{CLAYxFeO}) + 0.009(\text{CLAYxSILT}) + 25.846]$$

To use these equations:

1) Estimate clay content (%)
2) Estimate silt content (%)
3) Estimate total ion content of the leachate (%): This is the sum of soluble salts and iron in solution.
4) Estimate free iron oxide content (%).
5) Estimate the pore velocity of the leachate (cm/day). (The Darcian velocity divided by the leachate filled porosity.)
6) Determine the particular relative concentration.
7) Substitute the quantities from steps 1-5 into the equation determined from step 6.

For example, suppose the soil has the characteristics:

Clay content = 15%, silt content = 20%, free iron oxide content = 1.5%. Suppose the leachate has total soluble ions 0.08%. If the porosity of the soil was 0.38 and the infiltration rate was 0.6 cm^3/cm^2/day, then the pore velocity would be 0.6/.38 = 1.6 cm/day. Suppose the initial cadmium concentration in the leachate is 2 ppm and the concentration limit was 1 ppm. Then the relative concentration of interest is 0.5. Substitution into the equation for V .5 yields

$$\begin{aligned} V_{.5} = \frac{1.6}{30}[&-1.289(15) + 0.016\,(15 \times 15) + \\ &1.212\,(0.08 \times 20) - 10.245\,(1.5) + \\ &0.149\,(15 \times 1.5) + 0.010\,(15 \times 20) + \\ &+ 28.391] \\ = {}&0.290 \text{ cm/day} \end{aligned}$$

To summarize this point, we have plotted the data from all the columns run and have matched theoretical curves to the observed curves to get individual values for D, K_1 and K_2 for each column. At this point the only data that have been used are values of α, v, c_0 and the experimental breakthrough curve from the column. And for selected relative concentrations we have migration rates for each column.

To generalize these results from the few columns run to a more general setting and also to display the results in an easily accessible form we make use of multiple regression equations. One regression equation for each of the selected relative concentrations.

LIMITATIONS AND ADVANTAGES OF THIS MODEL APPROACH

Limitations: Caution must be exercised when using this approximation for several reasons. First, these were based on column experiments and not field conditions. Second, although the regression of velocity with both soil and leachate properties have an r^2 value on the order of 0.82, considerable variation in the actual velocity values and those predicted by the regression equations are still present. These variations are most noticeable in those soils which are low in both clay and silt. Here we feel that these regression equations should not be relied upon for highly accurate predictions of Cd movement. Rather they should be used to gain some idea of the degree to which the selected soil and leachate properties either contribute to or detract from the Cd transport velocity.

Refinements in the Model are needed in order of most increase in predictive capability per expenditure of a given amount of development work. These include:

1) Tests with leachates radically different from those used to date.
2) Inclusion of more soil and leachate parameters.
3) Use of more than 15 reference points on curves (possibly 21; that means much greater computer costs, but also means greater predictable precision).
4) More analyses of sensitivity of output to character of input so it can be determined where to exercise most care in collecting input data and which experimental conditions to control most closely in developing the Model such as
    --importance of soil bulk density and pore size distribution.
    --nature of total organic carbon.
    --identification of the effect of other hydrous oxides such as Mn and Al.
    --importance of soil-leachate matrix pH.

Advantages: The Model provides certain advantages over others:

1) Less complicated than many other models.

2) Does not require infinite information of attenuation mechanisms which would need many years to discover and evaluate.

3) D, K_1, and K_2 have been derived as real values from research experimental data making it possible to apply the model to actual landfill disposal problems for attenuation of potentially hazardous metals. In short, D, K_1, and K_2 represent real values beyond armchair theory.

4) The model is most likely to be applied in its present form to the evaluation of disposal sites because it is

--independent of computers, as far as the ultimate user is concerned,

--much easier to use the products in the form of mathematical equations or tables,

--less demanding for data than other prominent simulation model approaches,

--no more limited by the quality of input data than other approaches, and (for a given site) much less limited by quantity of input data than other approaches,

--most likely to be used in evaluating relative migration of a single metal at several alternative sites (because of extrapolation problems, no claim of predicting exact movement rate in undisturbed soil is made. However, the model is very likely to estimate accurately metal movement rate in disturbed soils as might be found when landfills are lined with clay-soil mixtures or with native soils).

5) Concerned with nonconservative solutes (such as metals) and nonconventional leachates as well as raw, unspiked municipal solid waste leachates.

6) Not conflicting with the large hydrology modeling programs of the USGS.

7) Conveniently and accurately related to easily measurable physical and chemical properties of the soil-leachate system.

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ENVIRONMENTAL IMPACT OF ALTERNATIVE METHODS OF LANDFILLING ON SURFACE WATER AND GROUND WATER

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ABSTRACT

Suitable landfill sites must be utilized efficiently. Methods to maximize site utilization include solid waste volume reduction (milling and baling), increased site height (hillfills), and use of existing excavations (coal strip mines). In this study, these methods are being evaluated and compared with a standard sanitary landfill to determine their environmental impact, especially on surface and ground-water resources.

A balefill, millfill, hillfill, coal strip mine fill and permitted sanitary landfill have been selected with similar physical and climatological settings. All available data were collected pertaining to operational history, climatology, hydrology, geology, soils, and surface and ground-water quality. Data were assimilated and additional water quality monitoring points were installed. Water quality samples are now being collected.

INTRODUCTION

Solid waste generation and disposal represent a continuing environmental problem throughout the United States. The amount of solid waste being disposed on the land represents 90 percent of the total municipal solid waste. This solid waste is disposed in approximately 15,000 sites of which approximately one-third are permitted facilities. The Resource Conservation and Recovery Act (RCRA) requires the phasing out of open dumps within the next five years which will result in the disposal of increased volumes in existing sanitary landfills and/or the opening of large landfills.

Suitable landfill sites must be utilized efficiently. Where suitable sites are available, there will be emphasis on placing more refuse in the site by increasing its height, i.e., hillfills, or through volume reduction, i.e., milling or baling. In addition, large abandoned strip mines offer potential for the resolution of waste disposal problems from major metropolitan areas. The growing need to completely utilize the space available in suitable waste disposal sites through volume reduction or use of alternate methods necessitates an evaluation and comparison with the standard sanitary landfill to determine their environmental impact, especially on surface and ground-water resources.

The alternative methods of waste disposal (millfill, balefill, hillfill, and strip mine landfill) represent potential pollution sources as does the standard sanitary landfill. At this time, however, the characteristics of these facilities that promote pollution or conversely prevent it, have not been fully documented.

Each method has characteristics that affect their pollution potential. Milled refuse decomposes rapidly producing large quantities of concentrated leachate. Millfills however stabilize in a shorter period of time with reduced concentrations and volumes of leachate. Balefills, conversely, are expected to produce less concentrated leachate for a long period of time due to the high degree of compaction. Strip mine landfills present interesting and somewhat unique problems. Pollution due to acid mine drainage is common throughout mining regions of the United States. The added pollution potential of solid waste in these strip mines combined with the complex hydrogeologic system can create serious pollution problems. Hillfills are also unique in that large volumes of waste are disposed above the ground surface. Leachate breakout between lifts, gas production, and structural failure are among their inherent problems as well as the potential for large volumes of leachate.

In order to determine the environmental impacts of each type of landfill on water resources, a study is being conducted which will evaluate and compare volume reduction and alternative methods of waste disposal to sanitary landfills. In addition the study will also determine site characteristics that control the occurrence or non-occurrence of leachate. These data obtained, through the study of each type of landfill, will be utilized to extrapolate each method's usefulness into other geographic areas.

SITE SELECTION

Site selection was a critical element of this study since there are many variables (depth to water, geology, hydrology, refuse-processing, and site preparation). Site selection criteria were therefore established and included:

1. Availability of background data.
2. Availability of detailed engineering and scientific reports.
3. Availability of the site.
4. Accessibility of the site.
5. Status of litigation, if any.
6. Overall representative nature of the site.
7. The site must be unlined.
8. The refuse must have reached field capacity and be generating leachate.
9. All sites must have similar climatology.

The site selection criteria were then applied to each type of landfill located east of Mississippi River (see Table 1). This included an evaluation of 15 balefills, 23 millfills, 17 hillfills, and 253 landfills in coal strip mines. Upon selection of the five initial landfills for study, field evaluations were made at each of the selected landfill sites to confirm the existing data, assess the physical setting, and conduct geologic and hydrologic evaluations as necessary. Sources of meteorological data, additional data pertaining to the operational history, and data from the regulatory agencies were collected during the field evaluation. It was found, after the initial field evaluation, that several of the sites did not meet the criteria and additional sites were selected.

TABLE 1. Alternative Methods of Solid Waste Disposal in Selected States East of the Mississippi River

STATE	Balefill	Millfill	Hillfill	Coal Strip Mine Fill
Alabama	2	0	0	0
Connecticut	0	0	0	0
Delaware	0	1	0	0
Florida	0	3	1	0
Georgia	2	2	0	0
Illinois	0	1	6	40
Indiana	0	0	3	6
Kentucky	0	0	0	15
Louisiana	0	2	0	0
Maine	1	1	0	0
Maryland	0	1	0	2
Massachusetts	1	1	0	0
Michigan	0	0	1	0
Minnesota	1	0	2	0
Mississippi	0	0	0	0
Missouri	0	0	0	6
New Hampshire	0	0	0	0
New Jersey	1	1	0	0
New York	6	1	0	0
North Carolina	0	1	0	0
Ohio	1	2	0	56
Pennsylvania	0	0	1	120
Rhode Island	0	0	0	0
South Carolina	0	4	1	0
Tennessee	0	0	0	2
Vermont	0	0	0	0
Virginia	0	0	1	0
West Virginia	0	0	0	6
Wisconsin	0	2	1	0
Total	15	23	17	253

Upon final selection of the five sites for the study, data gathered during the field evaluation together with other background data collected in earlier stages of the project were evaluated in order to determine the possible needs for additional monitoring. The monitoring system was to delimit ground and surface water flows and their interrelationships as well as to develop or refine water table maps and limiting flow lines. Existing chemical data were analyzed to assist in the placement of additional monitoring points at each site.

SELECTED SITES

Millfill

The millfill, located in the northeastern United States, has been in operation since 1973 receiving an average of 285 tons of refuse per day. The landfill operates on a 130-acre tract of which 45 acres have been filled (see Figure 1). The fill is up to 50 feet thick.

The site is located on the lower slope of a valley wall. It is underlain by coarse, poorly-sorted glacial deposits which are used for daily cover. Ground water is at a depth from 5 to 15 feet below ground surface and discharges to a nearby major river to the south of the site. The ground water has been monitored by five wells and one spring since the beginning of the operation. Two additional wells were emplaced in the fall of 1978 to augment the existing monitoring network. Collection of representative ground-water samples is made after pumping each well dry and allowing it to recover. A grab sample is obtained from the spring.

Balefill

The balefill is located in the north-central United States and was in operation for approximately two years, receiving an estimated 238,000 tons of baled refuse during that period. The landfill utilized 39 acres during that period (see Figure 2). The fill is up to 30 feet thick.

The site is located on a gently-rolling upland area. It is underlain by extremely coarse, poorly-sorted glacial deposits which are used in part for daily cover. Ground water is at an average depth of 200 feet below ground surface and flows in an eastward direction to a major river. Ground water at the site had not been monitored prior to this study. Leachate from the site had been collected and analyzed. Four wells were installed during the summer of 1978 to implement a ground-water monitoring system. In addition to these wells one additional background well and two leachate collection points are monitored for this study. Representative ground-water samples are collected after pumping at least two well volumes from each well. Grab samples are obtained from the leachate sampling points.

Hillfill

The hillfill is located in the north-central United States and operated between 1965 and 1976 receiving an estimated 750 tons of refuse per day. This site is approximately 150 feet high and occupies an area of 39 acres.

The site is located on a glacial plain adjacent to several streams (see Figure 3). This site was constructed utilizing relatively impermeable clay from on-site excavations. Daily cover was provided by both the clay and glacial till. Ground water is at a depth from 5 to 26 feet and discharges to adjacent streams. The ground water at the site has been monitored since 1969 utilizing nine monitoring wells and several adjacent water supply wells. Surface water samples

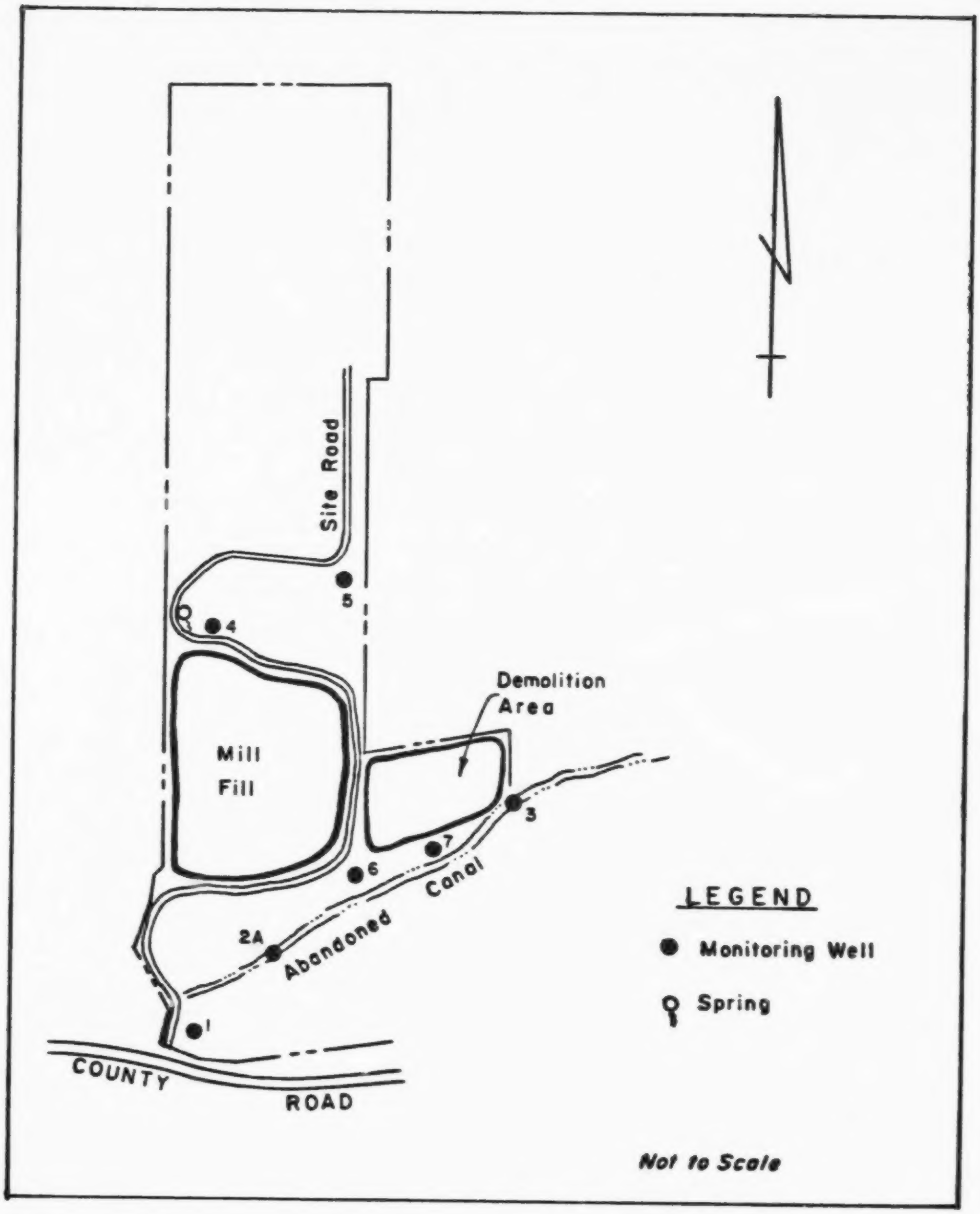
Site Road
5
4
Mill Fill
Demolition Area
3
7
6
Canal
2A
Abandoned
1
COUNTY ROAD
LEGEND
● Monitoring Well
♀ Spring
Not to Scale

Figure 1. Millfill site plan and location of monitoring points.

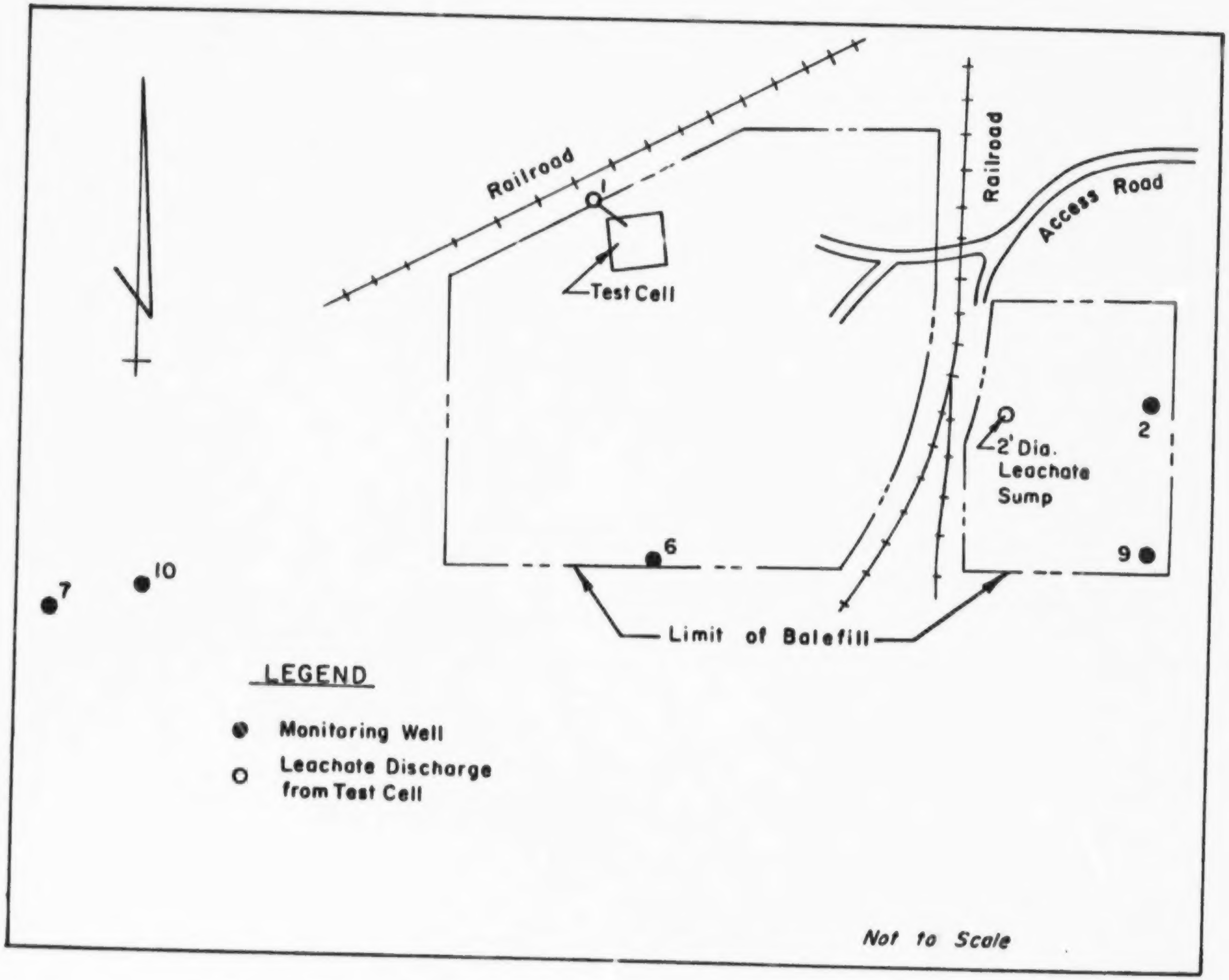
Railroad
1
Test Cell
Railroad
Access Road
2' Dia.
Leachate
Sump
2
9
6
10
7
Limit of Balefill
LEGEND
● Monitoring Well
○ Leachate Discharge from Test Cell
Not to Scale

Figure 2. Balefill site plan and location of monitoring points.

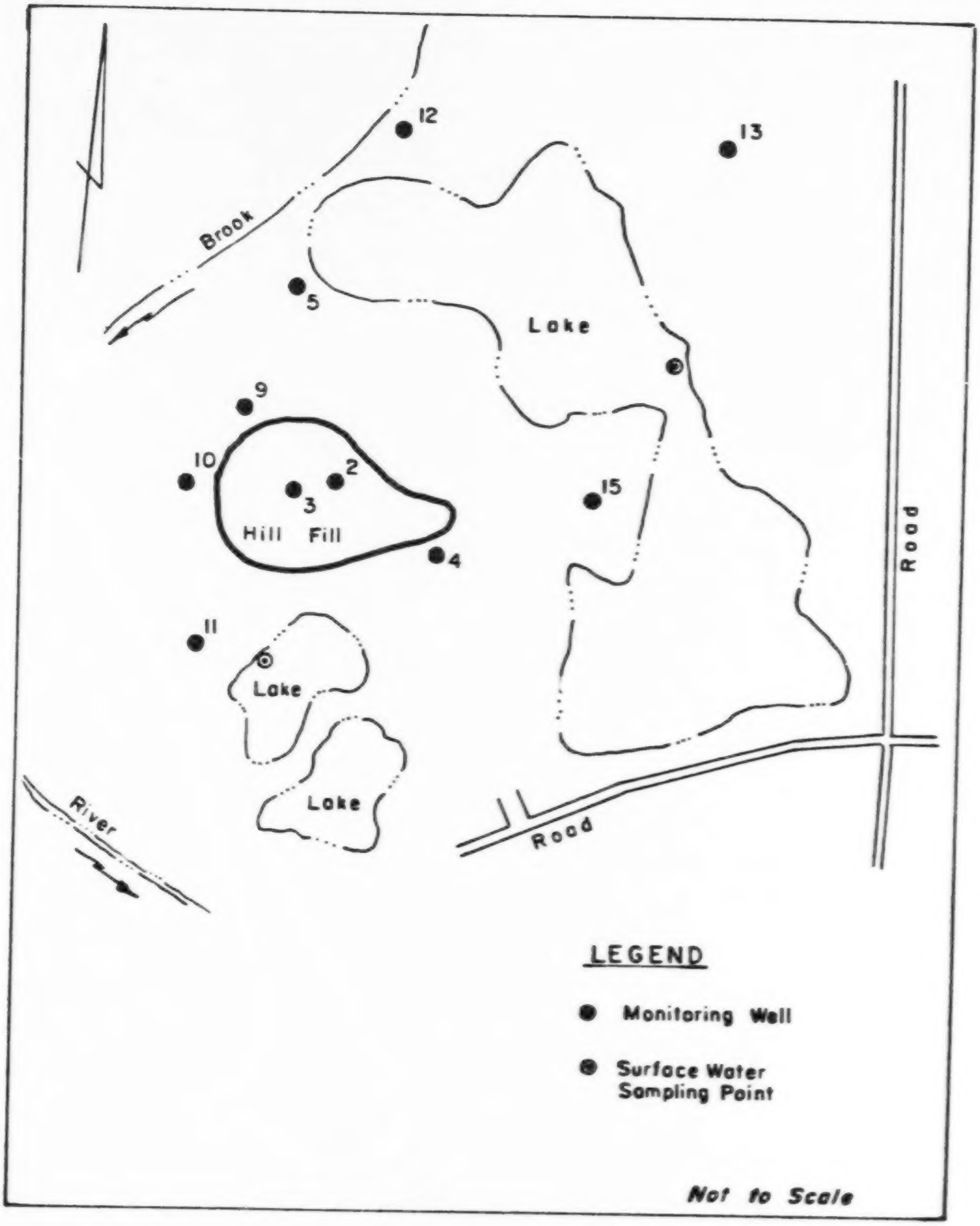
12
13
Brook
5
Lake
9
10
2
3
Hill Fill
15
4
Road
11
Lake
Lake
River
Road
LEGEND
● Monitoring Well
⊛ Surface Water Sampling Point
Not to Scale

Figure 3. Hillfill site plan and location of monitoring points.

have also been collected from the adjacent lakes and streams. Two additional wells were emplaced during the fall of 1978 to augment the existing monitoring network. Representative groundwater samples are collected after removing two well volumes from the wells by bailing. Grab samples were obtained from the lakes.

Coal Strip Mine Landfill

The coal strip mine landfill is located in the northeastern United States and has been in operation since 1968 receiving an estimated 165 tons of refuse per day. The landfill operates on a 40-acre tract of which an estimated 20 acres have been filled (see Figure 4). The fill is up to 80 feet thick.

The site is located near the top of a hill which had previously been strip mined. Daily cover is strip mine spoil. Ground water at the site varies from a depth of several feet immediately below the operation to 126 feet at the ridge top above the operation. Ground water at the site previously had been monitored through use of one spring and a hand-dug well. Surface water was also monitored at this site. Three wells were emplaced during the summer of 1978 to monitor ground-water conditions. These wells in addition to the existing sampling points are sampled. Representative ground-water samples are collected by removing at least two well volumes from each of the wells prior to sampling. A grab sample is obtained from the spring, the hand-dug well, and the stream.

Permitted Sanitary Landfill

The landfill is located in the northeastern United States and was operated between 1971 and 1975 using the trench method. It accepted an estimated 112 cubic yards of domestic refuse and demolition debris daily. The landfill utilized a total of 22 acres with waste being deposited in thicknesses up to 20 feet (see Figure 5).

The site is located on a topographic and hydrologic divide. The area around the landfill slopes steeply down toward two intersecting creeks. Twenty to forty feet of poorly-sorted, stoney clay till underlies the site. Bedrock is a gray, dense, low permeability shale. The till was used for both intermediate and final cover. Ground water is 40 to 50 feet below ground surface and discharges to the adjacent creeks. The ground water has been monitored by six wells completed around the landfill. Two stream sampling points have also been monitored during and after the period of operation. Each well is pumped dry and allowed to recover before collection of ground-water samples in order to assure representative samples. Grab samples are obtained from the surface water monitoring point.

SAMPLE COLLECTION - MONITORING

Selection of chemical parameters for analysis was based on the materials deposited at the site and from the interpretation of existing data. From these two sources, it was determined which ions are most likely to be migrating from the site and will be useful indicators of contamination. The following chemical analyses are being performed on each sample: alkalinity, acidity, sulfate, chlorides, total solids, total dissolved solids, nitrate nitrogen, amonia nitrogen, total phosphate, total kjeldahl nitrogen, copper, iron, manganese, sodium, lead, zinc, total organic carbon, COD. In addition pH, specific conductivity, and temperature are measured in the field when the samples are collected. Samples are being

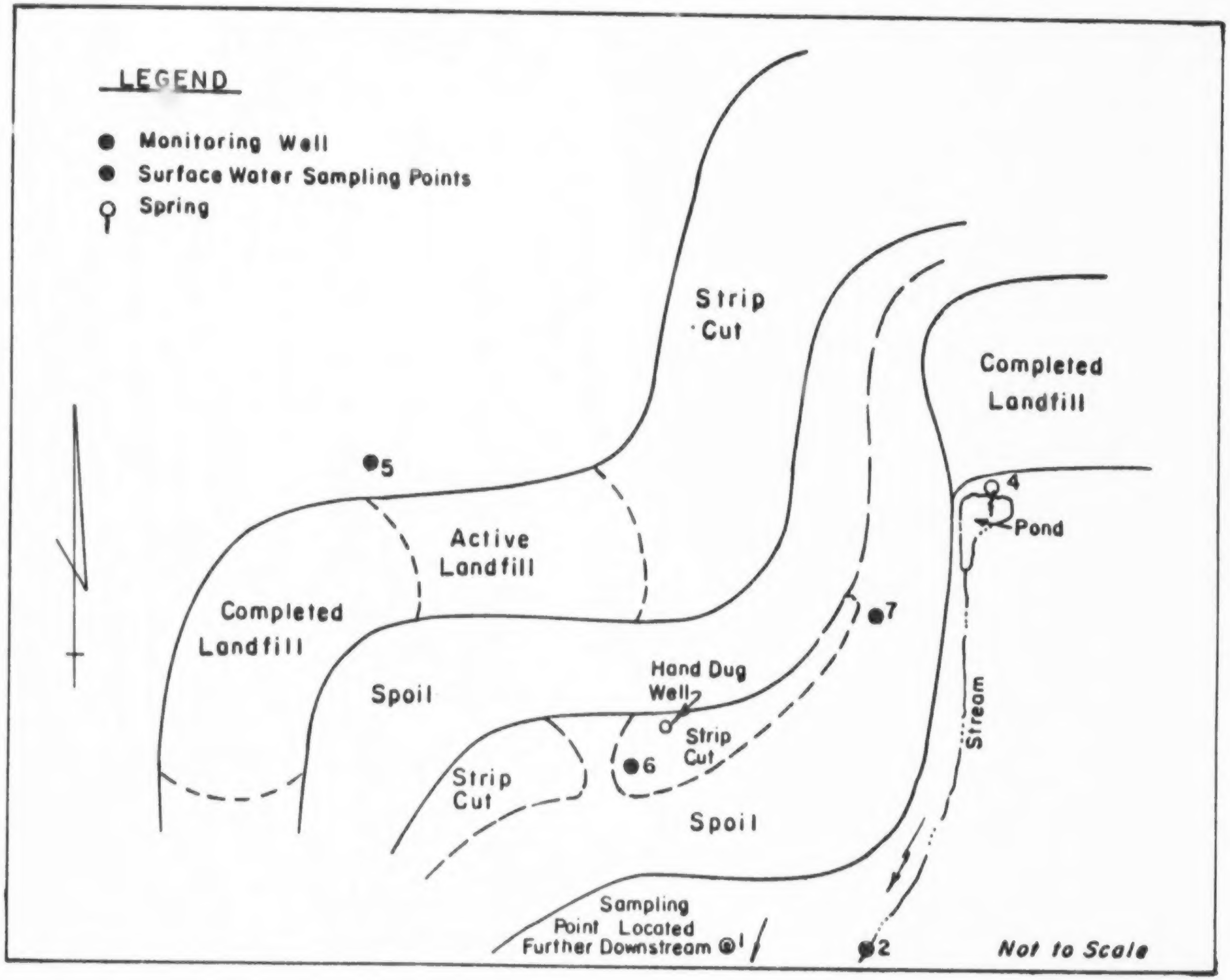
LEGEND
● Monitoring Well
● Surface Water Sampling Points
Spring
Strip Cut
Completed Landfill
5
Active Landfill
Completed Landfill
Spoil
4
Pond
7
Hand Dug Well
Strip Cut
6
Strip Cut
Spoil
Stream
Sampling Point Located Further Downstream 1
2
Not to Scale

Figure 4. Coal strip mine landfill site plan and location of monitoring points.

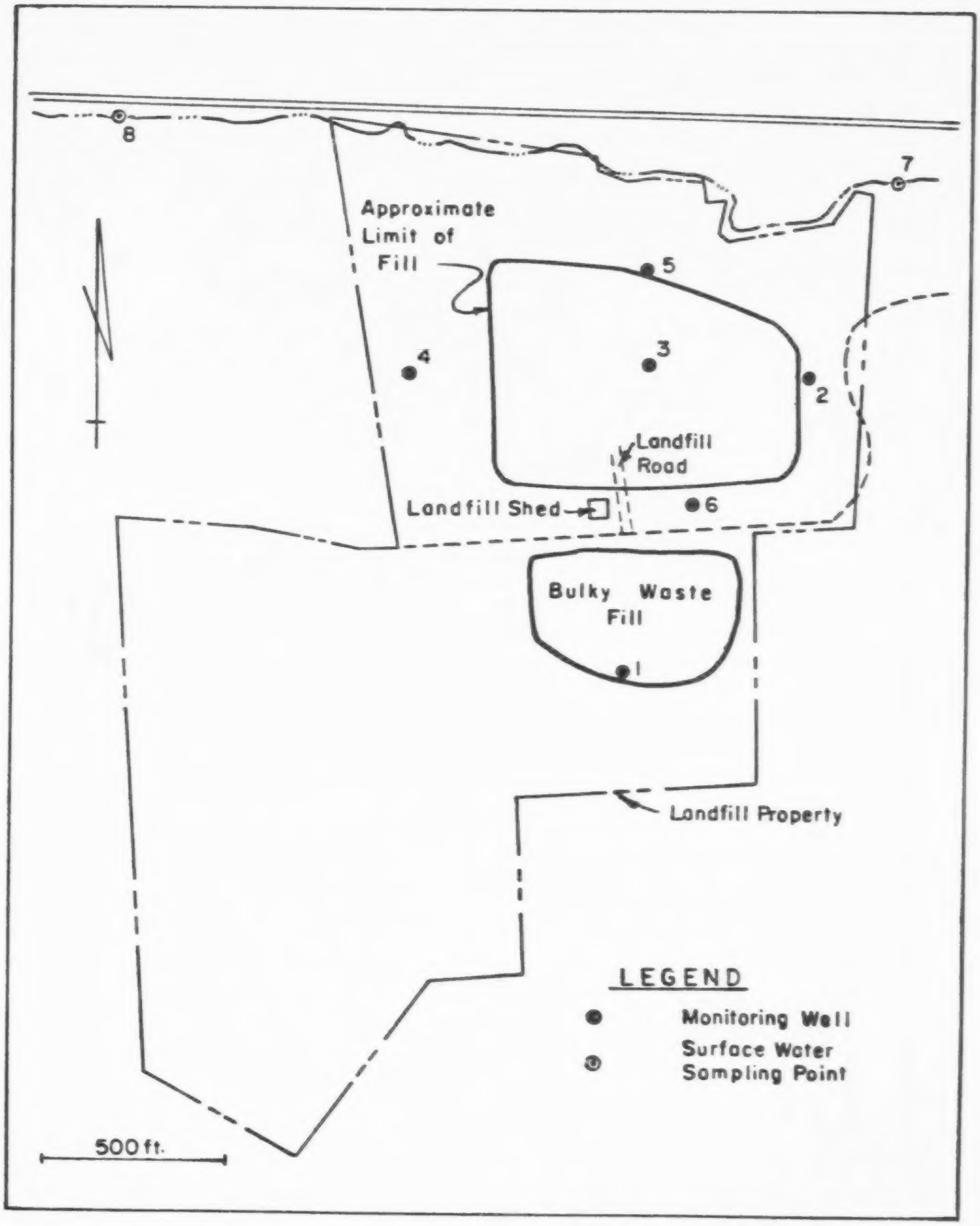
8
7
Approximate
Limit of
Fill
5
3
4
2
Landfill
Road
Landfill Shed
6
Bulky Waste
Fill
1
Landfill Property
LEGEND
● Monitoring Well
⊙ Surface Water Sampling Point
500 ft.

Figure 5. Permitted sanitary landfill site plan and location of monitoring points.

collected quarterly over a 12-month period at each of the five selected sites. Each sample is coarse-filtered in the field, preserved in accordance with EPA recommended methods, and shipped to a private laboratory for analysis within 24 hours of sample collection.

All chemical and hydrologic data are being stored in a computer to insure rapid and accurate recall. As needed, statistical tests will be applied to the data to determine variations occurring between sites.

In order to assess the effectiveness and usefulness of each of the five disposal methods, it will be necessary to: (1) reduce both historical and current chemical data to a meaningful format; (2) use hydrologic and meteorological data to develop a water budget; and (3) determine the pollution potential and possible areas of applicability for each method.

The impact of each site on the environment is a function of the quality and quantity of leachate generated, the overall attenuation characteristics of the site, and hydrogeolic regime at the site. The type of landfill operation is expected to control the rate and strength of leachate generation; the siting and engineering of each disposal site will govern the attenuation and hydrogeologic effects.

CONCLUSIONS

Data about millfills, hillfills, balefills, and landfills and coal strip mines have been collected throughout the eastern United States. Site selection criteria have been developed and utilized in the selection of five sites. Background data at each of these five sites have been collected and the existing monitoring network evaluated. Additional monitoring points were located where necessary to augment the existing network. The first set of samples from all sites was collected late fall of 1978. Sample collection will continue on a quarterly basis for one year.

ACKNOWLEDGEMENTS

The work which is reported herein is being performed under Contract #68-03-2620, Donald E. Sanning, Project Officer.

PREDICTING LANDFILL GAS MOVEMENTS IN SOIL AND EVALUATING CONTROL SYSTEMS

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INTRODUCTION

The Ohio State University has been involved in developing analytical tools to be used by the practicing engineering profession for the design of sanitary landfills to reduce methane migration potential. Figure 1 shows a flow chart delineating the major tasks and their sequencing. The details of these studies have been described in the publications listed in the bibliography.

The purpose of this paper is to present a case history involving prediction of methane migration at a proposed landfill site. The prediction will be based upon the use of the design charts published by Moore (1976) and by Moore and Jayko (in press); however, the site geology does not allow for using the charts in a direct manner. Rather, approximations are required.

DESIGN EXAMPLE

Figure 2 shows a hypothetical site plan for a landfill. The geologic stratigraphy (Figure 2a) consists of a partially saturated clay layer, the top of which is at elevation 500 feet. The clay layer is overlain by 50 feet of uniform sand, which is in turn overlain by 3 feet of partially saturated clay. The lower clay layer supports a perched water table rising to elevation 515 feet. Erosion has resulted in the formation of a bluff 23 feet high which will serve as one side of the landfill. The natural bluff slopes at 2 horizontal on 1 vertical.

The landfill will be placed adjacent to the bluff with its base coinciding with the erosion plane abutting the bluff (elev. 530). The top of the landfill will be at elevation 570 feet.

Approximately 310 feet east of the eastern edge of the landfill, a river has cut a valley creating a bluff extending to below elevation 500 feet. Approximately 100 feet west of the western edge of the landfill, a fault has resulted in sound limestone being uplifted to form the base of the erosion plane at elevation 530 feet.

In plan view, the landfill occupies 4.5 acres and is shaped as shown in Figure 2b.

It is desired to evalute the methane migration potential of the site by the use of design charts.

Life being what it is, the conditions at this site do not apparently coincide with the conditions assumed in developing the design charts published by Moore (1976) or Moore and Jayko (in press). Thus it will be necessary to use the charts in an approximate manner.

Initially, we must determine if the charts are appropriate to use at all. The charts were developed assuming that the flow of methane was purely diffusional; that is, the pressure within the landfill is atmospheric. Such is likely to be the case if the landfill is well vented, either through a reasonably pervious cover, or through artificial vents such as pipes. The charts were also developed based upon 70% mole fraction methane in the landfill for a period of 5 years, after which time the methane concentration in the landfill reduces to zero. Seventy percent methane represents an upper theoretical limit; whereas, 5 years decomposition time represents a relatively short decomposition time. Nevertheless, the current state of knowledge concerning methane generation within landfills suggests that the assumptions made in developing the charts are

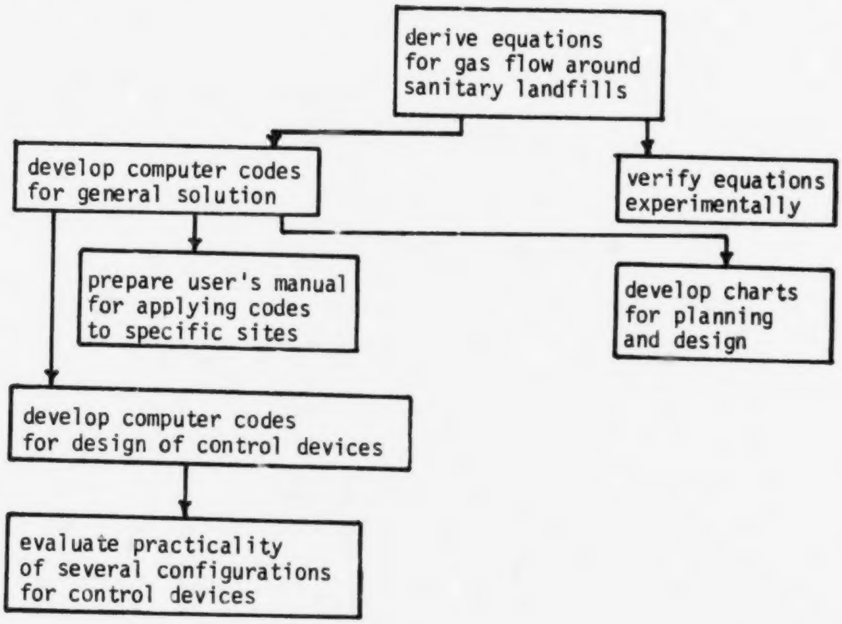
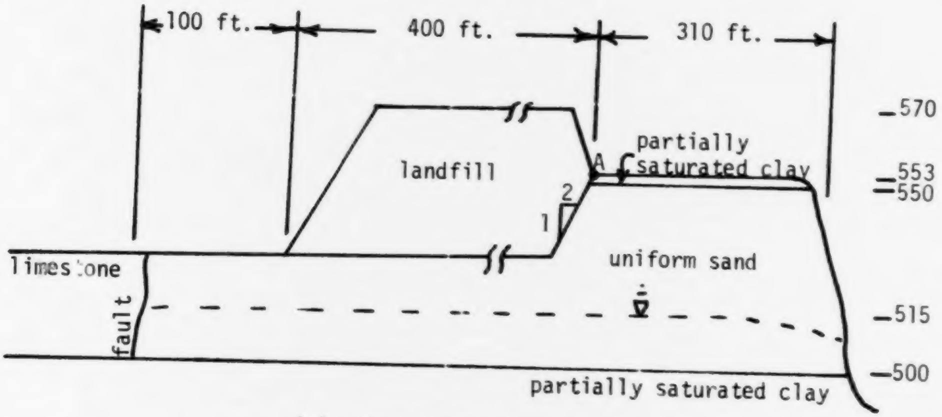
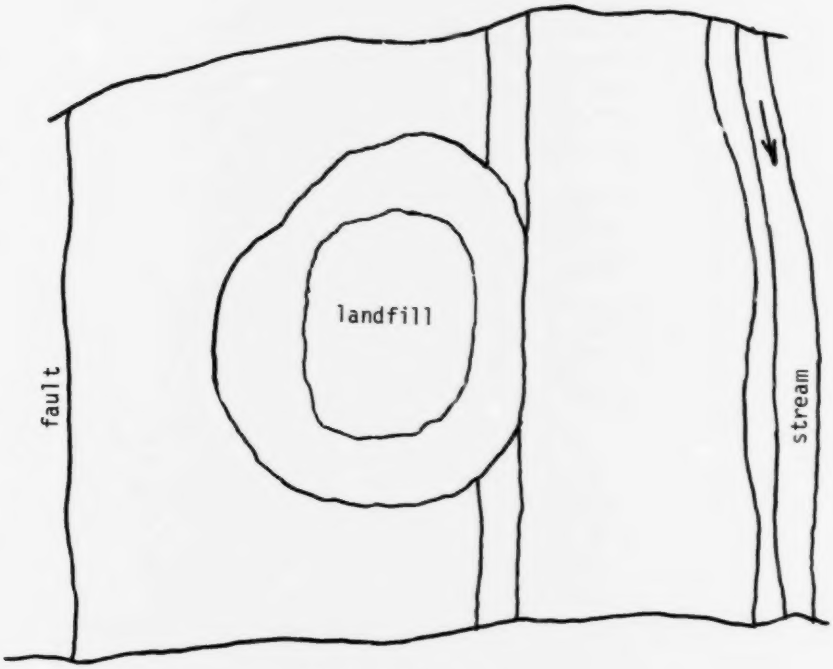

derive equations for gas flow around sanitary landfills
develop computer codes for general solution
verify equations experimentally
prepare user's manual for applying codes to specific sites
develop charts for planning and design
develop computer codes for design of control devices
evaluate practicality of several configurations for control devices

Figure 1 - Flow chart describing research program.


100 ft.
400 ft.
310 ft.
570
landfill
partially saturated clay
A
553
550
2
1
limestone
uniform sand
fault
515
500
partially saturated clay

(a) West-east crossection.


landfill
fault
stream

(b) Plan view

Figure 2 - Hypothetical landfill configuration.

appropriate for many U.S. landfills.

In order to apply the charts we must establish the geometric conditions and boundary conditions. Reference to Figure 2 shows that the site geometry is complex and does not comply with the simple geometry assumed in developing the design charts. However, it is possible to adapt the charts to suite the site geometry shown. The two sides (East and West) of the landfill are clearly dissimilar: The East side consists of sand capped with clay. Moreover, the landfill is located both above and below ground. The side of the landfill slopes, and a bluff exists of little over 300 feet from the landfill edge. The West side consists of uncapped sand, and the landfill is located entirely above grade. A limestone deposit is encountered 100 feet West of the edge of the landfill. Finally, in plan view the landfill is not circular.

To resolve this dilemma, one can make the following approximations:

One can calculate an equivalent radius of the landfill, r, by assuming a circular landfill having an area equal to that of the existing landfill:

$\pi(r)^2 = 4.5$ acres $= 2 \times 10^5$ square feet. Thus, $r = 250$ feet.

Now evaluate the methane migration distance on the East and West sides independently. Comments will be made later on the consequences of this action.

For the East side, one must now address the issue of the above grade portion of the landfill. This, however, is not a dilemma at all. Because the gas flow is diffusional only, that portion of the landfill above grade has no effect on the problem and can be ignored. Thus, the depth of the landfill should be taken as

$$d_f = 533 - 530 = 23 \text{ feet.}$$

The next problem is the sloping side of the landfill-soil interface. To circumvent this, one must assume that the edge of the landfill is located at the point of intersection of the landfill and the ground surface (point A in Figure 2). The landfill-soil interface will be assumed vertical; thus, in effect, a portion of the soil is assumed to be replaced by solid waste. The effect of this assumption will be to overestimate the methane migration distance somewhat. Thus, one can now determine the radius of the fill to be

$$r_f = r = 250 \text{ feet.}$$

One must now determine the radius to the soil boundary, r_s. Reference to Figure 2 shows that the bluff formed by the river valley constitutes a pervious boundary of zero methane concentration. The sloping bluff will be approximated as being vertical; thus, $r_s = 250 + 310 = 560$ feet.

The depth to an impervious boundary, d_s, will simply be the depth to the ground water table. (The effect of the depressed surface near the bluff is ignored). Thus,

$$d_s = 533 - 515 = 38 \text{ feet.}$$

Having defined the geometric conditions, it is now necessary to select the proper chart to apply to the problem. The soil is a sand and, therefore, coarse grained. The bluff at the river valley is in free contact with the atmosphere, and thus constitutes a pervious boundary. The conditions at the ground surface present an additional problem. The effect of the clay layer will be to impede venting of methane to the atmosphere, however, it does not constitute a perfectly impervious layer. Nevertheless, imperviousness will be assumed, recognizing that this approximation results in overestimating the distance of methane migration.

Thus, the appropriate chart to use is Figure A2 from Moore and Jayko (in press) which is reproduced in Figure 3. The values to be used in entering the chart are

$r_s = 560$ feet $\quad r_f = 250$ feet
$d_s = 38$ feet $\quad d_f = 23$ feet

$$\frac{d_s}{d_f} = \frac{38}{23} = 1.65$$

$$\frac{r_s}{r_f} = \frac{560}{250} = 2.24.$$

The chart is entered at $d_s/d_f = 1.65$. We then move vertically until an interpolated curve for $r_s/r_f = 2.24$ is encountered. Finally, we move horizontally to obtain $r_t/r_f = 1.88$. Thus, the maximum distance of migration of the 5% methane contour will

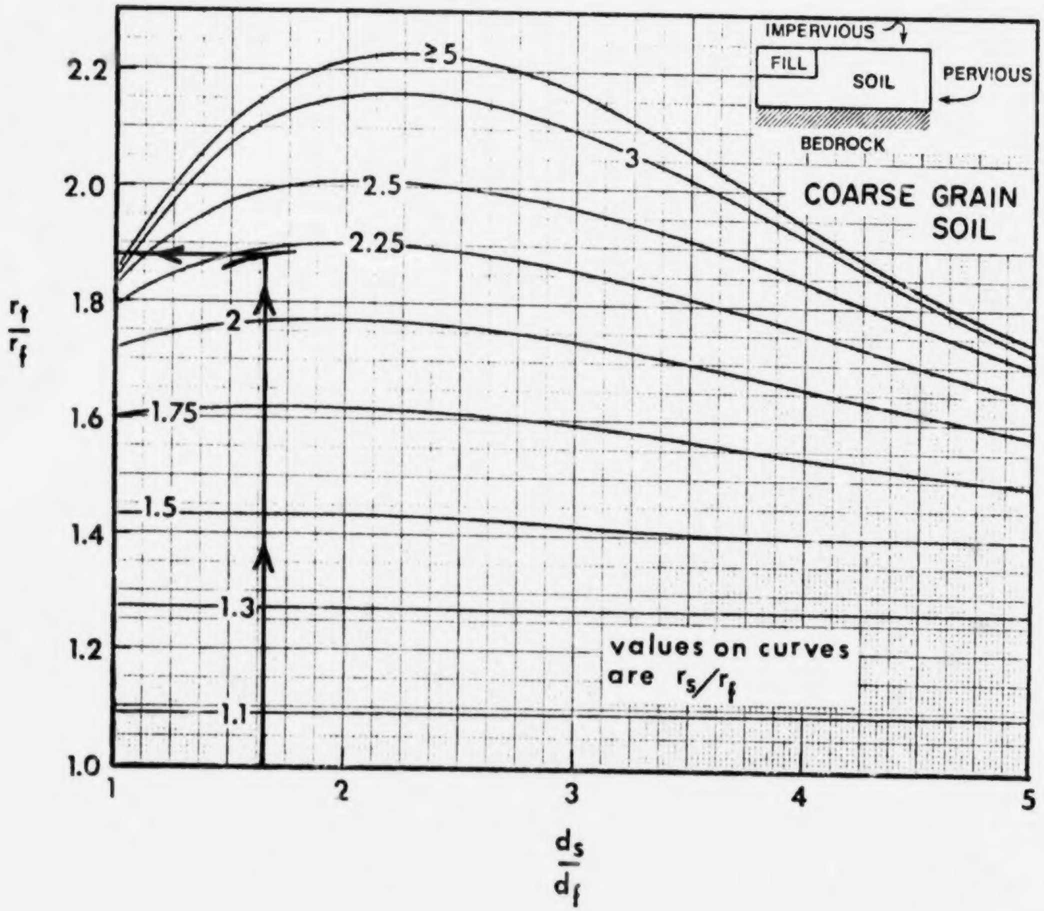


Figure 3 Design chart for granular soil (impervious ground surface and pervious radial boundary). From Moore and Jayko (in press).

be

$$r_t = 1.88\ r_f = 1.88\ (250) = 470 \text{ feet.}$$

This places the 5% contour extent at 470 - 250 = 220 feet beyond the edge of the landfill.

We now turn to the west side of the landfill. Here the landfill exists entirely above grade. As with the east side, the portion above the ground is ignored, therefore $d_f = 0$.

The radius of the fill is simply $r_f = r = 250$ feet. The radius to the soil boundary will be taken as $r_s = 250 + 100 = 350$ feet under the assumption that the sound limestone encountered at the fault constitutes a gas impervious boundary.

As on the east side, the depth to an impervious layer is taken as the depth to the groundwater table. Thus $d_s = 530 - 515 = 15$ feet.

Having defined the geometric conditions, it is now necessary to choose an appropriate chart. The soil is sandy and, therefore, coarse grained. The limestone constitutes an impervious radial boundary. The uncapped sand-atmosphere interface is pervious. Thus the appropriate chart is Figure A.3 from Moore and Jayko (in press) which is reproduced as Figure 4. The values to be used in entering the chart are

$r_s = 350$ feet $r_f = 250$ feet

$d_s = 15$ feet $d_f = 0$ feet

$$\frac{d_s}{d_f} = \frac{15}{0} = \infty$$

$$\frac{r_s}{r_f} = \frac{350}{250} = 1.4.$$

The value of $d_s/d_f = \infty$ could present a problem; however, reference to Figure 4 shows that for each value of r_s/r_f, the curves are essentially horizontal for d_s/d_f greater than approximately 4.0. Thus $d_s/d_f = 5.0$, the highest value on the chart, is an appropriate value to use for $d_s/d_f = \infty$. Entering the chart at $d_s/d_f = 5.0$, we move up to $r_s/r_f = 1.4$ (we lucked out, because there is a curve for 1.4, thus we need not interpolate) and then over to find $r_t/r_f = 1.242$. Thus the maximum distance of migration of the 5% methane contour will be

$$r_t = 1.242\ r_f = 1.242\ (250) = 310 \text{ feet.}$$

This places the 5% contour extent at 310 - 250 = 60 feet beyond the edge of the landfill.

Thus the 5% methane contour extends 220 feet to the east of the landfill and 60 feet to the west of the landfill.

One must now address the issue of the separate treatment of the two sides of the landfill. The concept upon which the design charts are abased involves exact symmetry around the landfill. Considering first the east side, the solution giving a migration distance of 220 feet would have presumed an equal migration distance on the west side. The fact that the actual migration distance on the west side is only 60 feet means that, as far as the east side is concerned, there is less methane on the west side than would be anticipated. Therefore the migration distance predicted for the east side based on the simplifying approximation will overestimate the actual condition somewhat.

The converse arguement for the west side would indicate that the simplifying assumption results in underestimating the actual gas migration distance on the west side.

Finally, one must discuss the consequences of assuming the landfill to be circular in plan view. Reference to figure 1b shows that in actuality the landfill is longer in a north-south direction than in the east-west direction. The assumed radius of 250 feet results in overestimating the methane migration distances in the east and in the west directions. Had estimates been made in the north and south directions (impossible, from a practical point of view), they would have underestimated the methane migration distance. Figure 5 shows a plan view delineating this author's approximation of the 5% methane migration contour around the landfill based upon the calculations performed above. The following factors influenced the decisions made in drawing the contour.

Directly to the east of the landfill, the basic distance was taken to be 220 feet

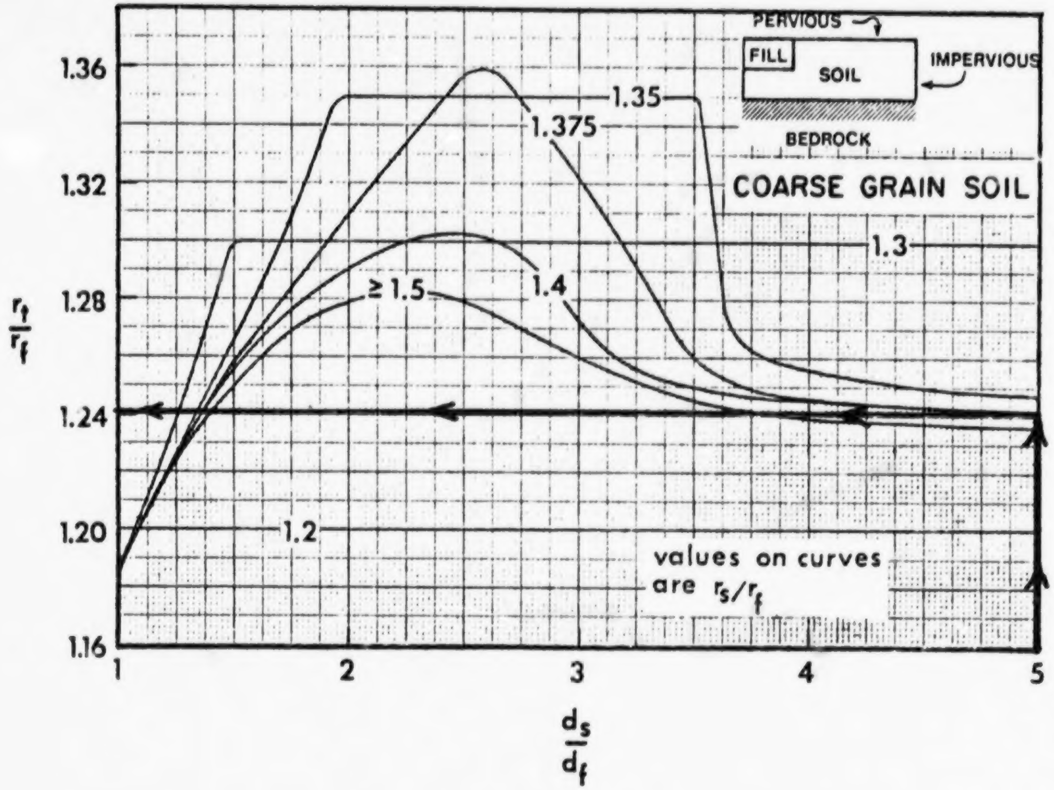


Figure 4 Design chart for granular soil (previous ground surface and radial boundary). From Moore and Jayko (in press).

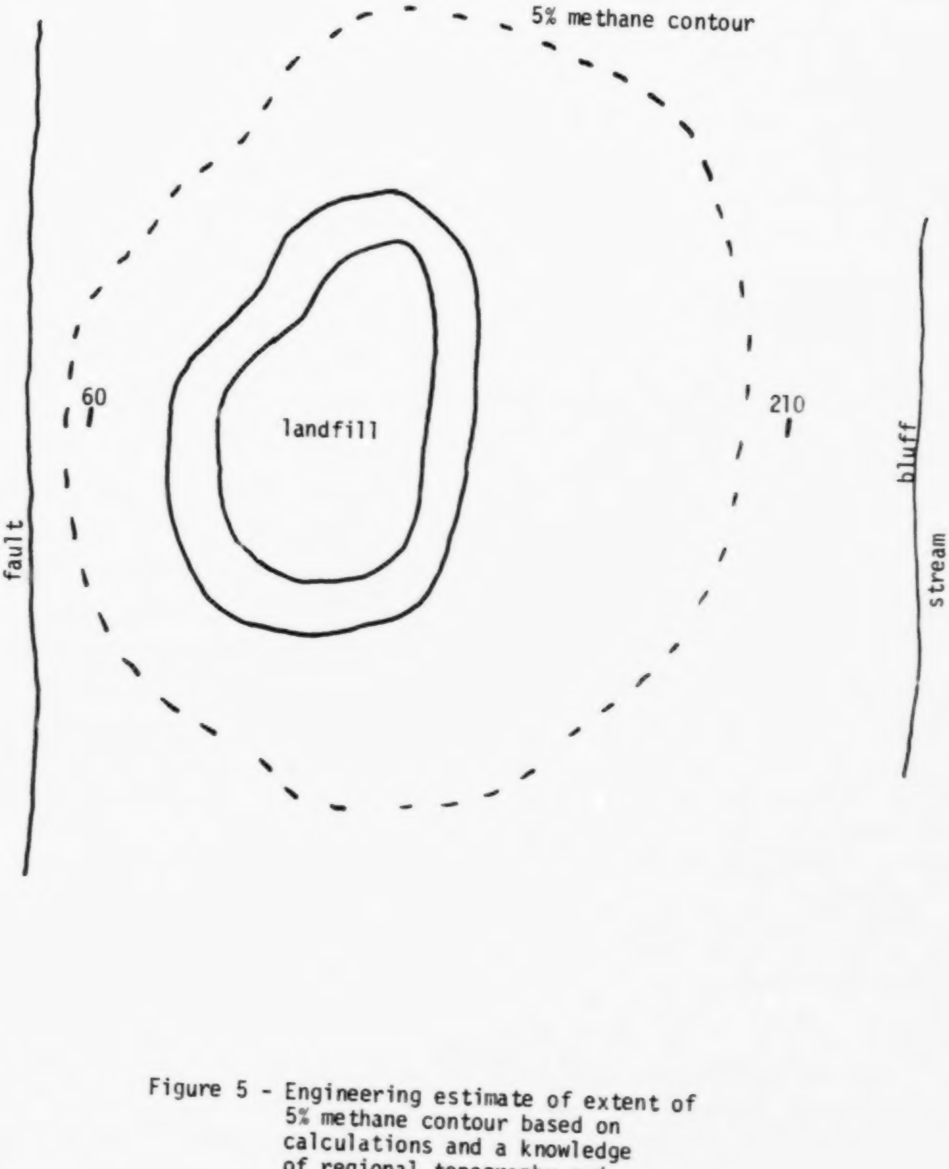

5% methane contour
60
210
landfill
fault
bluff
stream

Figure 5 - Engineering estimate of extent of 5% methane contour based on calculations and a knowledge of regional topography and geology.

from the edge of the fill. The following factors supported a reduction in this distance:

1. the clay cap is not totally impervious as assumed,
2. the clay cap was assumed to be infinitesimally thin; whereas, it is actually [illegible] feet thick,
3. the landfill is narrower in the east-west direction than assumed,
4. the methane migration distance on the west side is only 60 feet, and
5. the sloping landfill-soil interface was accomodated by replacing some soil by solid waste.

Directly to the west of the landfill, the basic distance was taken to be 60 feet from the edge of the fill. The following factor supported a reduction in this distance:

1. the landfill is narrower in the east-west direction than assumed.

The following factor supported an increase in this distance:

1. the methane migration distance on the east side is 220 feet (considerable greater than 60 feet).

The factor dictating increasing the 60 foot distance on the west side was judged to take precedence over the factor suggesting a decrease. Increasing the distance from 60 to 70 feet was pure engineering intuition. The author would not be at all surprised if field measurements disclosed that the 5% methane level migrated all the way over to the limestone.

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GAS MIGRATION AND MODELING

by

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ABSTRACT

A research program is currently underway to measure pressures and gas compositions both in a landfill and at its periphery, and to use the results to estimate gas production rates in the landfill and to calibrate a gas migration model.

Gas production rates will be estimated by monitoring gas compositions and pressures before, during and after application of a zone of negative pressure within the landfill. This zone is provided by means of a gas withdrawal well installed vertically in the landfill and vented to a pump. Results from a trial pumping test are presented.

Several sets of pressure and concentration measurements have been obtained at the migration study location. A typical set of data is presented, and shows the influence and effectiveness of a forced venting system in operation at the site. The data from all migration studies will be used to examine seasonal trends in migration patterns, and to calibrate a gas transport model. An example of the model's ability to simulate field conditions is given.

The influence of atmospheric pressure on gage pressure readings is demonstrated. It is suggested that barometric pressures in the field be obtained at the same time as gage pressures, and that all pressures be converted to absolute pressures for purposes of comparison.

1. INTRODUCTION

The decomposition of organic matter in the absence of oxygen results in part in the production of gaseous by-products. Under conditions usually encountered in sanitary landfills, these gases consist primarily of methane (CH_4) and carbon dioxide (CO_2). Small amounts of ammonia (NH_3), hydrogen sulphide (H_2S), nitrogen (N_2), hydrogen (H_2) and carbon monoxide (CO) can also be produced (Farquhar and Rovers, 1973).

Except in one or two isolated incidences, gas production in sanitary landfills received little attention until the early 1970's. Recently however, the potential hazards of these gases at landfills have become a major concern. While some recognition has been given to the acidification of groundwaters from the dissolution of CO_2 as a problem with respect to water quality, the major focus of attention has been directed toward CH_4 and the hazards presented by combustion.

To date there have been many cases reported where CH_4 produced in sanitary landfills has migrated into adjacent properties and has created explosive conditions when in concentrations ranging between 5% and 15% volume in air. Where explosions have occurred, property damage has been substantial. In some cases there has been personal injury, even the loss of life.

In response to this danger, regulatory agencies have begun to recommend the installation of systems for venting or diverting migrating gases. In some cases these measures have been effective. In others, they have not. The problem appears to be that the patterns of gas production and migration are poorly understood. This stems in part from an incomplete knowledge of the properties of the gas as produced in the landfill and the inability to anticipate the manner and extent to which it migrates in the soil.

A research program conducted by Waterloo Engineering Associates Inc. in conjunction with the University of Waterloo was begun in April, 1978 to address these problems. The scope of this investigation and the results obtained to date are presented below.

2. OBJECTIVES

The objectives of the investigation are

1. To study the extent of methane production in landfills with respect to
   - a) gas composition,
   - b) gas pressure and
   - c) gas flow rates;
2. To measure the patterns of gas movement into soils adjacent to landfills;
3. To calibrate a previously developed gas flux model for a specific landfill site; and
4. To demonstrate the model's capability to predict gas concentrations in the soil
   - a) under typical variations in climatic conditions and
   - b) for various gas control mechanisms such as vents, diversions and confinements.

These objectives are being pursued through implementation of the experimental program described in subsequent sections.

3. GAS PRODUCTION IN LANDFILLS

3.1 Scope of Experimental Work

The objective of this phase of the work is to provide information about gas composition, pressure and production in sanitary landfills. Such information is essential to both the analysis of gas migration into adjacent soils and the assessment of gas recovery potential at landfills. Of the three measurements, gas production is the most difficult. Most investigations of this property have been on a laboratory scale. Actual field measurements are sparse.

This study is examining the extent of landfill gas production and its variation through the review of published literature and the collection of field data. Estimates of gas production rates will be obtained by monitoring gas compositions and pressures before, during and after application of a zone of negative pressure within the landfill. This zone is provided by means of a gas withdrawal well installed vertically in the landfill and vented to a pump.

3.2 Literature Review

Estimates of gas generation rates for landfills given in the literature vary widely, from values of 2.2 l/kg dry weight (Coe, 1970) to 250 l/kg dry weight (Mandeville, 1976; De Walle *et al.*, 1978. The characteristics of landfill sites often vary considerably, so that field studies must be conducted to determine the actual production rate at a given site.

Several field studies similar to the one described herein have been conducted to determine gas production rates. As in this study, gas was removed from the landfill at various flow rates via a gas withdrawal well, and gas production rates were estimated based on the steady-state flow rate.

Gas generation rates determined at four landfills in California are reported in Table 1. It can be seen that gas generation rates varied between 22 and 45 ml/kg-day. DeWalle *et al.* (1978), using sealed containers filled with different amounts of solid waste and maintained at different environmental conditions, obtained average gas production rates over a 300 day period ranging from 0.29 to 18.1 ml/kg-day. Based on these experiments, they concluded that a gas production rate of 20 ml/kg-day can be obtained at most landfills, with higher or lower production rates obtained with higher or lower moisture contents. Their suggested rate of 20 ml/kg-day is comparable with the rates reported in Table 1.

Blanchet (1976) suggested that the optimum gas withdrawal rate from pumping wells is 0.09 m^3/min per m of well. This rate is comparable to the rates reported for the last three landfills in Table 1.

Table 1 GAS PRODUCTION RATES AT CALIFORNIA LANDFILLS

	Palos Verdes	Mission Canyon	Sheldon-Arleta	Mountain View
Depth of Well (m)	33.5	30.	38.	12.
Radius of Influence (m)	76.	-	76.	40.
Gas Flow Rate (m^3/min)	8.5	2.3	4.1	1.4
Dry Density (kg/m^3)	714.	-	595.	595.
Gas Production Rate				
a) ml/kg-day	30.	-	22.	45.
b) m^3/min per m of well	0.25	0.08	0.11	0.12
Reference	Schuyler, 1973	Schuyler, 1973	City of Los Angeles, 1975	E.P.A., 1977

3.3 Instrumentation and Monitoring of Site

The three-dimensional nature of the negative pressure zone caused by pumping from the gas withdrawal well is measured with nests of gas sampling piezometers located on three radii from the gas withdrawal well, each nest consisting of five piezometers at various depths. A plan view of the gas production site is shown in Figure 1. Piezometer nests 1 through 8 have been installed to date.

The gas withdrawal well is constructed of 4" PVC pipe, with the bottom 3.3 m of the pipe perforated and wrapped with fibreglass tape to provide screening.

A typical piezometer nest installation is shown in Figure 2. The bottom piezometer is constructed of 1/2" PVC pipe while the other piezometers are 1/4" polyethylene tubing. The tubing is attached to the PVC pipe which provides a relatively rigid central core and aids in the installation of the piezometer nest. A length of 1/4" tubing is also connected to the top of the 1/2" PVC pipe. Each piezometer is perforated over the bottom 0.6 m and wrapped with fibreglass tape.

The piezometers are surrounded by pea gravel in the monitoring wells, and separated by concrete plugs. Each of the 1/4" tubes are of different colours, with each colour corresponding to a relative depth in a well. Use of coloured tubing provides a means of instant recognition of piezometer placement, and prevents errors caused by monitoring incorrect depths.

The tubing is routed to a central location at the gas withdrawal well through buried trenches. Approximately 2 m of the tubing leading from each piezometer is coiled around the gas withdrawal well within a partially buried 0.6 m diameter corrugated metal pipe with a removable locked top. This provides a secure arrangement for storage of the tubing and allows all piezometers to be monitored from a single location.

At the commencement of a pumping test, each piezometer is connected via the 1/4" tubing to a monitoring board, as shown in Figure 3. Each tube has a separate on/off valve to allow the piezometers to be sealed in the intervals between measurements. The monitoring board is also equipped with a

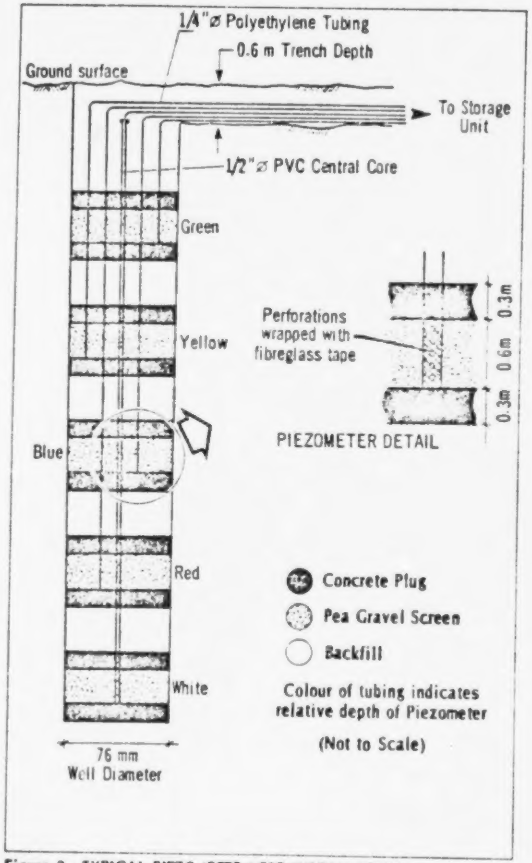
1/4"⌀ Polyethylene Tubing
0.6 m Trench Depth
Ground surface
To Storage Unit
1/2"⌀ PVC Central Core
Green
Yellow
Blue
Red
White
76 mm Well Diameter
Perforations wrapped with fibreglass tape
0.3m
0.6m
0.3m
PIEZOMETER DETAIL
Concrete Plug
Pea Gravel Screen
Backfill
Colour of tubing indicates relative depth of Piezometer
(Not to Scale)

Figure 2 – TYPICAL PIEZOMETER NEST INSTALLATION

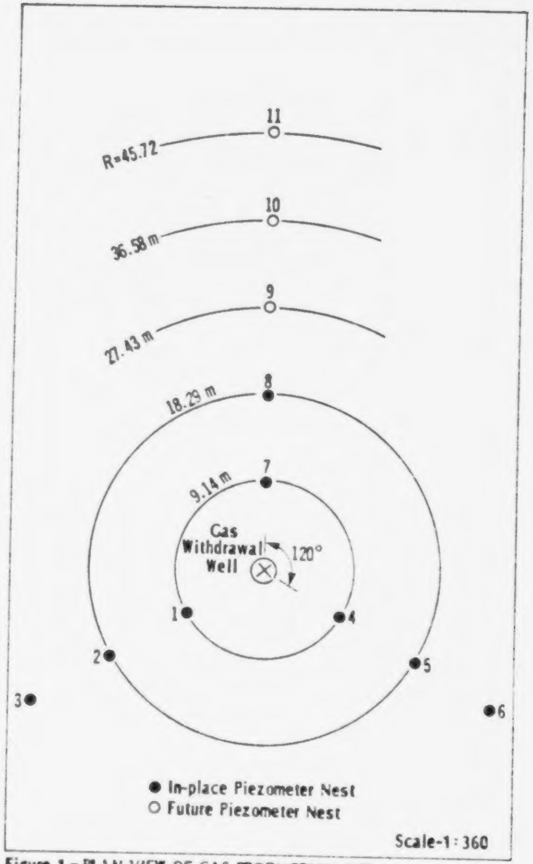
R=45.72
36.58 m
27.43 m
18.29 m
9.14 m
11
10
9
8
7
1
2
3
4
5
6
Gas Withdrawal Well
120°
● In-place Piezometer Nest
○ Future Piezometer Nest
Scale-1:360

Figure 1 – PLAN VIEW OF GAS PRODUCTION SITE

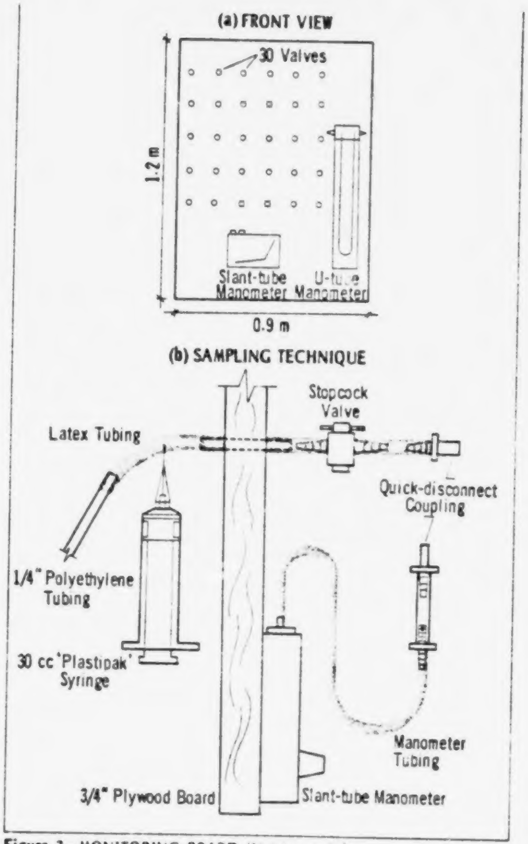
(a) FRONT VIEW
30 Valves
1.2 m
Slant-tube Manometer
U-tube Manometer
0.9 m
(b) SAMPLING TECHNIQUE
Stopcock Valve
Latex Tubing
Quick-disconnect Coupling
1/4" Polyethylene Tubing
30 cc 'Plastipak' Syringe
Manometer Tubing
3/4" Plywood Board
Slant-tube Manometer

Figure 3 – MONITORING BOARD (Not to scale)

slant tube manometer which can read pressures (positive or negative) from 0.01 to 3 inches of water. Pressures are obtained by connecting the appropriate valve to the manometer and turning the valve on. Such an arrangement allows pressures in the landfill to be monitored very quickly, even for a large number of piezometers.

Gas concentration samples are collected in 'PLASTIPAK' disposable syringes, 30 cc capacity, with 'Luer-Lok' tips. First, a length of tubing leading to a piezometer is purged by applying an aspirator to the appropriate opened valve. Then the valve is closed, and the needle of a syringe is inserted into the latex tubing at the rear of the monitoring board. A sample is withdrawn, and the plunger in the syringe is allowed to equilibrate before the needle is removed from the tube. Difficulties are sometimes encountered when attempts are made to withdraw samples from a tube under a large negative pressure. The negative pressure pulls the plunger back into the syringe and prevents removal of a sample of gas.

The syringe needle is then removed and quickly replaced with a rubber serum stopper. Since the stoppers are self-sealing, samples required for gas analyses in the laboratory can be removed from a sample syringe by means of a 1 cc syringe inserted in the stopper.

Control experiments using the 'PLASTIPAK' syringes have shown that an insignificant amount of leakage occurs during the interval from sample collection to analysis. To ensure acquisition of reliable results, analyses are conducted in the laboratory as soon as the samples are returned from the field.

Samples are analyzed in the laboratory using a Fisher Model 1200 Gas Partitioner. Gases analyzed include methane, nitrogen, oxygen and carbon dioxide.

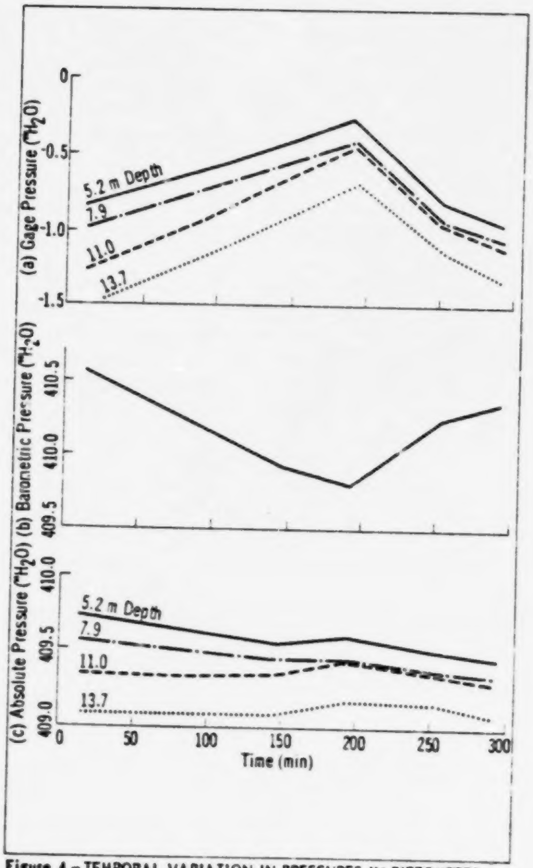

Figure 4 - TEMPORAL VARIATION IN PRESSURES IN PIEZOMETER NEST #6 (NOV 26, 1978)

3.4 Absolute Versus Gage Pressure

Problems were encountered with respect to interpretation of the pressure readings in the landfill after completion of a preliminary investigation of the gas production site on November 13, 1978. It was observed that pressures at individual piezometers displayed considerable variation over the three hour sampling period, with pressures ranging from approximately -1 to +0.5 inches of water, as measured by the slant tube manometer. It was concluded that pressure readings were being affected by the atmospheric pressure; i.e., even if the pressure in a piezometer remained constant, changes in atmospheric pressure would affect pressure readings on the manometer.

Another series of pressure readings were obtained over a subsequent five hour period on November 26, 1978, and barometric pressure at the site was also recorded during this interval. The results from one of the six piezometer nests are shown as a representative indication of the effect of barometric pressure on manometer readings. Results from the other five piezometer nests were similar.

The gage pressures in the piezometers measured with the manometer are shown in Figure 4 (a). Barometric pressure readings during the sampling interval are shown in Figure 4 (b). Absolute pressures (the sum of the barometric and gage pressures) are indicated in Figure 4 (c). It can be seen that while relatively large fluctuations in barometric pressure were observed, absolute pressures in the landfill remained fairly constant over the five hour period.

Absolute pressures were found to vary considerably from week to week, indicating that the landfill does respond to changes in atmospheric pressure but at a very slow rate.

Thus relying on gage pressures alone to give an indication of pressure changes within a landfill can lead to erroneous conculsions. For this reason, all subsequent pressure measurements were converted to absolute pressures.

3.5 Pumping Test

A trial pumping test was conducted at the gas production site on January 12, 1979. Gas concentrations and pressures were measured at the piezometers before initiation of pumping, and pressures were continuously monitored throughout the pumping phase of the test and after pumping ceased. The pump was allowed to run for 90 minutes at a rate of approximately 1.13 m^3/min, and pressure recovery within the landfill after cessation of pumping was monitored for 30 minutes.

Contours of methane concentrations in the monitored area before commencement of pumping are shown in Figure 5. The lowest methane concentrations were observed near the bottom of the refuse, which is located at an average depth of approximately 13.5m. During drilling of the monitoring wells, it was found that refuse at the bottom of the landfill consisted mainly of nonputrescible materials, which would explain the low methane concentrations. The shape of the contours near the surface of the landfill are uncertain. It has been assumed in Figure 5 that the contours are approximately orthogonal to the surface because the frozen soil presents an impermeable upper boundary. However it is possible that contours are approximately parallel to the surface very near the bottom of the frozen soil layer. Attempts will be made in future tests to determine upper contour configurations.

Gas samples were not collected during or after pumping for the trial test, but will be obtained in future tests.

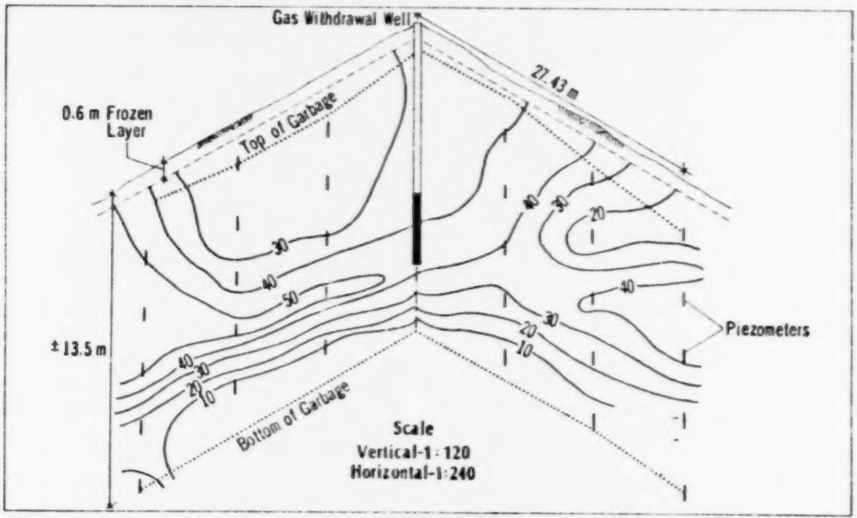

Figure 5 - INITIAL CONTOURS OF METHANE CONCENTRATIONS (%) (JAN 12, 1979)

Drawdowns in absolute pressure at piezometer nest 4 are shown in Figure 6. Measurements were not available for the top piezometer as it was plugged. As shown in Figure 6, the largest drawdowns were obtained in the piezometers located at the same depth as the gas withdrawal well screen, which was approximately 9m from piezometer nest 4. It appears that steady-state conditions were close to being achieved when the pump was shut off, however this will be verified in future tests involving longer pumping times.

It also appeared that pressures reached an equilibrium throughout the depth of the piezometer nest after pumping ceased. This is apparent in Figure 6, which shows that drawdown pressure 30 minutes after cessation of pumping was approximately -0.7 inches of water in each of the four piezometers.

Pressure contours in the landfill at the end of the 90 minute pumping period are shown in Figure 7. Atmospheric pressure at this time was approximately 409.9 inches of water, while pressures in the landfill near the gas withdrawal well screen were on the order of 408.8 inches of water. It is quite apparent that the zone of influence of the pump extended beyond the range of the monitoring wells. It is not possible at this time to estimate how far this zone of influence extends; monitoring wells 9 to 11 indicated in Figure 1 will be used to examine its extent.

Future pumping tests will involve running the pump at different discharge rates to attempt to contain the zone of influence within the range of our monitoring piezometers. Gas concentrations during pumping at the positive pressure end of the pump will also be obtained, as well as in the piezometers during the recovery period after pumping ceases. These measurements will be used to obtain estimates of gas production rates in the landfill, as explained in the following section.

3.6 Analysis Methodology

Results of the pumping tests described above will be used to estimate methane gas production rates in the landfill. Two methods will be used to obtain these estimates.

The first method will involve measuring gas concentrations exhausting from the pump after steady-state conditions have been achieved. The rate of production will then be estimated knowing the steady-state gas concentrations and the discharge rate and zone of influence of the pump. This technique was used in the four California studies listed in Table 1.

A second method of estimating methane production rates will be attempted to provide an alternative means of estimation and to allow results from the two techniques to be compared. However unlike the first method, which has been tried at several landfills with reasonable success, the second technique is still in the theoretical stage, and it is not known whether or not it will produce satisfactory results.

While the first method will involve monitoring gas concentrations and pressures during pumping with steady-state conditions in effect, the second method will involve monitoring concentrations and pressures during recovery of the landfill. After pumping ceases, pressure equilibration will occur in the zone of influence through the introduction of air from the gas withdrawal well and perhaps also the surface of the landfill. This equilibration is expected to occur fairly rapidly, and the results of the first pumping test and those conducted using a similar pumping technique at another landfill (Gartner Lee Assoc. Ltd., 1978) have indicated that this is likely to occur.

Thus it is anticipated that, within a short time after pumping ceases, pressures in the landfill will have completely recovered while methane concentrations will be partially depleted within the zone of influence of the pump. Gas composition in this zone will be monitored for some time, and the rate of methane production will be estimated based on the rate of recovery of methane concentrations and changes in the volumetric extent of the recovery zone.

4. GAS MIGRATION FROM LANDFILLS

4.1 Scope of Experimental Work

The objectives of this phase of the work are to measure the patterns of gas movement into soils adjacent to a landfill,

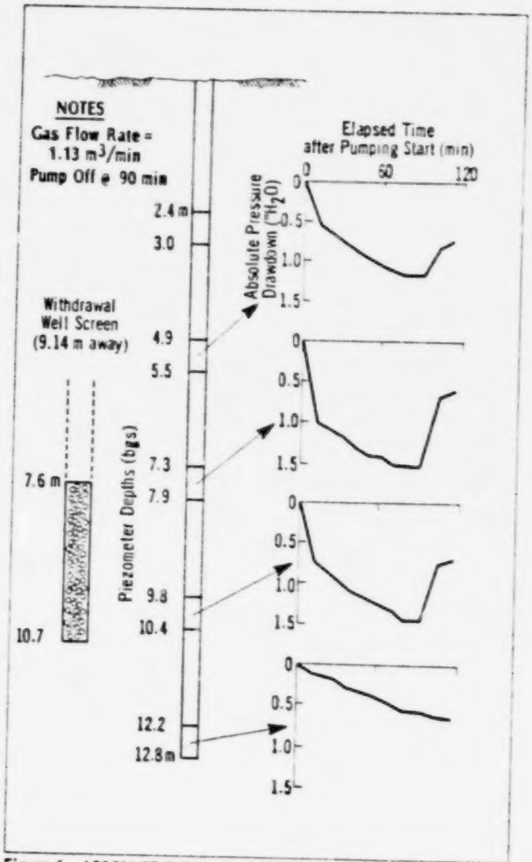

Figure 6 – ABSOLUTE PRESSURE DRAWDOWNS IN PIEZOMETER NEST #4 DURING PUMPING TEST (JAN 12, 1979)

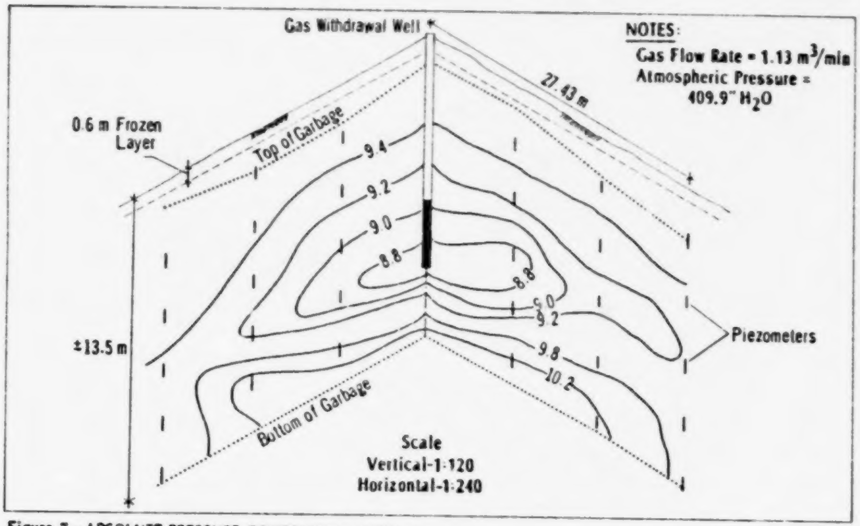
Gas Withdrawal Well
NOTES:
Gas Flow Rate = 1.13 m³/min
Atmospheric Pressure = 409.9" H2O
27.43 m
0.6 m Frozen Layer
Top of Garbage
9.4
9.2
9.0
8.8
8.8
9.0
9.2
9.8
10.2
±13.5 m
Piezometers
Bottom of Garbage
Scale
Vertical-1:120
Horizontal-1:240

Figure 7 – ABSOLUTE PRESSURE CONTOURS (+400" H_2O) AT END OF 90 min PUMPING PERIOD (JAN 12, 1979)

to examine the effects of precipitation and frost cover on these patterns, and to use these measurements to calibrate the gas flux model described in Section 5.

4.2 The Nature of Gas Migration

Gases generated in a landfill as a result of anaerobic decomposition of organic materials normally migrate upwards through the refuse and soil cover, and diffuse into the atmosphere. However when upward movement is impaired, the gases move laterally along subsurface paths of least resistance until they find a vertical path to the atmosphere.

Two subsurface conditions tend to promote lateral gas migration: those that restrict or prevent upward movement, and those that provide alternative routes for lateral movement. Vertical passage can be hindered by the soil cover compacted over the refuse, particularly when saturated with water or frozen. Surface structures such as asphalt pavement, concrete foundations and floor slabs can also confine gases and promote lateral migration. Alternative routes for gas migration are provided by manmade structures such as subserface utility structures and sewers, and also by cracks, fissures and voids resulting from landfill differential settlement.

Lateral migration of gases can occur to considerable distances beyond fill boundaries. In studies conducted on sanitary landfills in the Los Angeles area, methane concentrations of 10-20% were detected as far as 180 m from the boundaries of completed landfills at a depth of 0.6 m below the ground surface (County of Los Angeles, 1972). The extent of gas migration beyond the landfill boundaries is a direct function of soil permeability and of gas pressure and concentration within the landfill. Gases generated in a landfill surrounded by clay soils can travel only a short distance from the edges of the landfill. However in landfills surrounded by more permeable materials, such as those constructed in gravel quarries, the gases can travel a large distance from the fill boundaries.

Migrating gases consist mainly of carbon dioxide and methane. Methane in volumetric concentrations of 5-15% forms an explosive mixture with air. Duffusion of this mixture into enclosed buildings can create severe explosion and fire hazards.

Attempts to solve the problem of methane migration are usually by either the installation of impermeable barriers or ventilation systems around the perimeter of the landfill. Venting may be either natural or forced.

4.3 Instrumentation and Monitoring of Site

Changes in gas composition and pressure in the migration study area are being monitored by a number of piezometers, with the main line of monitoring wells positioned orthoganally to the edge of the refuse and extending approximately 50 m into the soil adjacent to the landfill. Two additional sets of wells are positioned on either side of the main line to examine spatial variations in pressures and gas concentrations along the perimeter of the refuse. A plan view of the gas migration study area is shown in Figure 8.

Piezometer installation and sample collection were conducted using the techniques described above for the gas production study.

4.4 Migration Studies Conducted to Date

Absolute pressures in the piezometers at the migration site were measured once a week for five weeks, commencing January 10, 1979. Gas compositions in the piezometers were also measured on three of these days. The results of January 17, 1979 are presented as an example of the data that were obtained.

Contours of average absolute pressures in inches of water and average percent methane concentrations for January 17 are shown in Figure 9. These contours represent average absolute pressures and concentrations throughout the depth of the migration zone, which extends to the groundwater table at a depth of approximately 8 m.

A forced venting system consisting of vertical perforated pipes extending to the groundwater table and connected to a horizontal header leading to a pump, is in operation at the migration study site. The influence of this system is shown by the

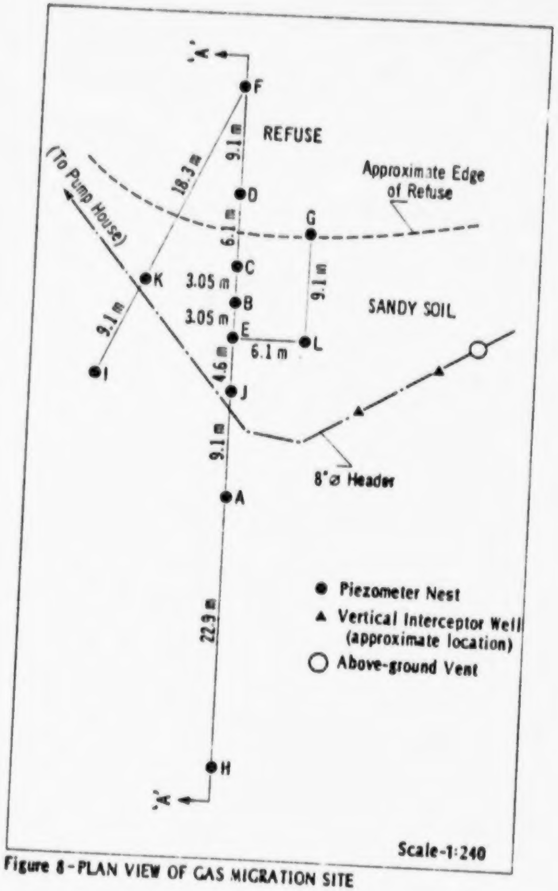
'A'
F
REFUSE
9.1 m
18.3 m
(To Pump House)
D
Approximate Edge of Refuse
G
6.1 m
C
K
3.05 m
B
9.1 m
SANDY SOIL
3.05 m
E
6.1 m
L
9.1 m
I
4.6 m
J
9.1 m
8" ⌀ Header
A
22.9 m
● Piezometer Nest
▲ Vertical Interceptor Well (approximate location)
○ Above-ground Vent
H
'A'
Scale-1:240

Figure 8 - PLAN VIEW OF GAS MIGRATION SITE

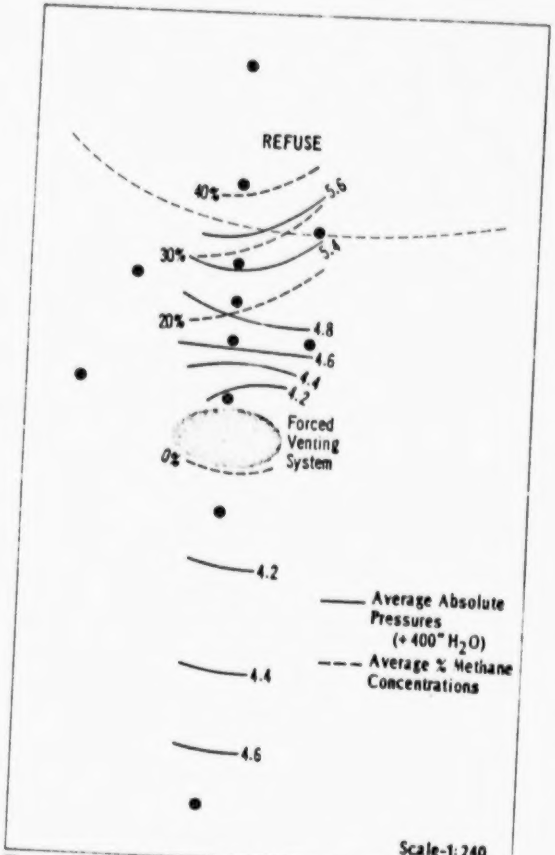
REFUSE
40%
5.6
30%
5.4
20%
4.8
4.6
4.4
4.2
Forced Venting System
0%
4.2
Average Absolute Pressures (+ 400" H2O)
Average % Methane Concentrations
4.4
4.6
Scale-1:240

Figure 9 - PLAN VIEW SHOWING CONTOURS OF PRESSURE AND CONCENTRATION AT GAS MIGRATION SITE (JAN 17, 1979)

magnitudes of the pressure contours in Figure 9. The effectiveness of the system is apparent from the methane concentration contours, which indicate that methane is not migrating beyond the venting system.

Variations in pressures and concentrations with depth on January 17 through the main line of monitoring wells, indicated as Section 'A-A' on Figure 8, are shown in Figure 10. The influence and effectiveness of the forced venting system is again apparent from these elevation views. Pressures appear to be fairly constant with depth in the vicinity of the venting system, indicating that gas is being withdrawn throughout the total length of the vertical perforated pipes. The large decrease in methane concentrations, from 50% in the refuse to 0% at the venting system, is also apparent in Figure 10. As with the production study, the shape of the pressure and concentration contours near the surface are uncertain at this time.

4.5 Analysis of Migration Results

Results from the other four sampling days were similar in some aspects to those reported for January 17. The results were similar in that approximately the same patterns of pressure and concentration contours were in effect on each of the study days. However, the magnitude of these contours displayed considerable variation from week to week, the most marked being those for absolute pressures. Changes in concentration contour magnitudes were caused by variations in methane concentrations in the refuse. No methane was found in the area beyond the venting system on any of the study days.

A number of additional migration studies will be conducted over the next several months. Of particular interest will be changes in migration patterns as thawing of the upper soil occurs.

The data from all migration studies will be used to examine seasonal trends in pressure and methane concentration patterns, and to calibrate the gas migration model described in the following section.

5. MODELLING METHANE MIGRATION

5.1 Model Description and Calibration

Data collected from the migration study site will be used to calibrate a previously developed model for gas transport in porous media (Mohsen et al., 1978). In the model, the diffusion-convection equations coupled with a mass conservation equation for a binary mixture of gases are

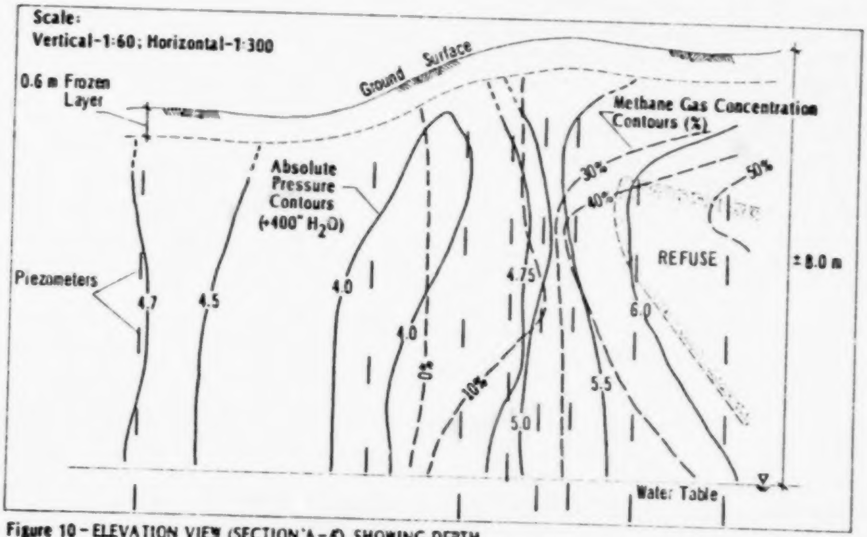

Figure 10 – ELEVATION VIEW (SECTION 'A–A') SHOWING DEPTH VARIATION IN PRESSURE AND CONCENTRATION AT GAS MIGRATION SITE (JAN 17, 1979)

solved using finite elements under a combination of Dirichlet, Neumann and Cauchy type boundary conditions. The finite element solution is a two step process. The first step solves for the pressure field, and mass and volume averaged velocities are computed from this field. The second step solves for gas concentrations at various points within the media. Density and viscosity changes are accounted for.

Field situations can be simulated using either an axisymmetric configuration with cylindrical coordinates, or a two-dimensional Cartesian system. The axisymmetric format consists of radial and normal (generally vertical) flux from the source. This is a realistic condition at many landfills. Although conditions must be uniform at a given radius, any number of horizontal soil strata is allowed. The axisymmetric format will accommodate various media and boundary configurations, both of which may be time-variant. For solutions close to the source, the model behaves like a two-dimensional orthogonal system.

In applying the model to the gas migration site, the main line of piezometer nests indicated as Section 'A-A' in Figure 8 will be discretized into a finite element configuration. Data requirements for application of the model to this location will include porosity, intrinsic permeability and moisture content for soils in the study section, and a deliniation of all relevant boundary conditions. In addition, relationships for viscosity and density as a function of gas concentration will be required and will be selected from the literature.

In calibrating the model, values of the relevant input parameters will be varied within their measured limits to give a best fit to field measured gas concentrations and pressures. The sensitivity of these parameters will also be examined.

5.2 Calibrated Model Applications

The calibrated model will be used to display variations in gas migration patterns and to evaluate various gas control methods.

The variety of gas migration patterns to be studied will include the following soil conditions:

a) unconfined from above, homogeneous, long term as normal;
b) statified soil including highly permeable layers as preferred migration routes;
c) confined from above to simulate winter conditions; and
d) time varying influent boundary conditions.

Methods for gas control at landfills will also be evaluated in terms of their effectiveness in reducing gas concentrations. The methods examined will include:

a) atmospheric vents as a function of penetration depth,
b) barriers, and
c) positive pressure systems for gas collection.

5.3 Example of Previous Application of Model

Model simulations have been performed on data collected from a previous investigation at the same landfill used for this study. The site configuration is shown in plan view in Figure 11. The figure also shows the existance of a venting trench along the northwest side of the landfill, as well as the overall dimensions of the trench and landfill. The forced venting systems indicated on the diagram were not in effect at the time this previous study was conducted.

A previous field study was conducted to evaluate the effectiveness of the trench in preventing the migration of methane gas. Figure 12 shows the pattern of multi-level gas sampling piezometers used in this previous investigation, and their positions relative to the trench. Samples were collected to determine gas compositions at the various piezometer locations, and these were used subsequently for comparison with the model simulation.

Cross-section 'B-B', as indicated in Figure 11 is shown in detail in Figure 13. This represents the radial configuration used for model simulation. Although data such as pressure contours, source properties and media characteristics were sparse, attempts were made to generate suitable values for the simulation process.

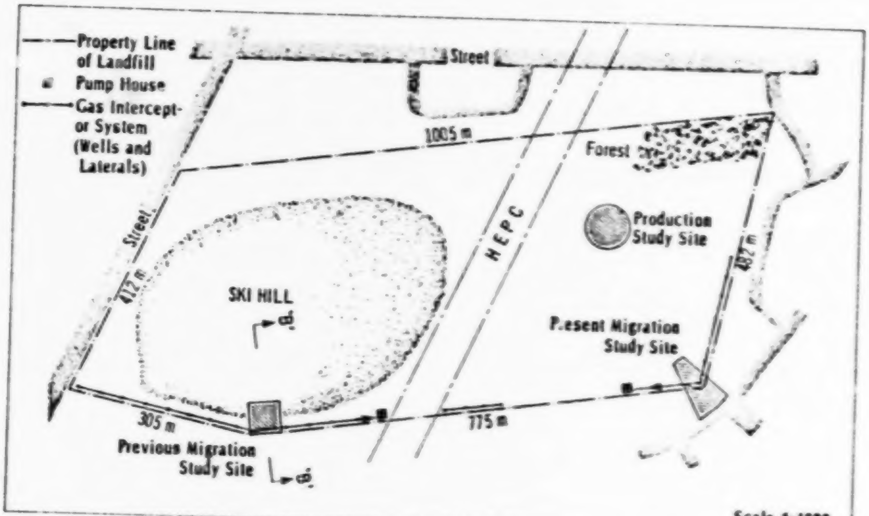
Property Line of Landfill
Pump House
Gas Interceptor System (Wells and Laterals)
Street
Street
1005 m
Forest
Production Study Site
HEPC
412 m
482 m
SKI HILL
Present Migration Study Site
305 m
775 m
Previous Migration Study Site
Scale-1:4800

Figure 11-LOCATION OF STUDY AREAS

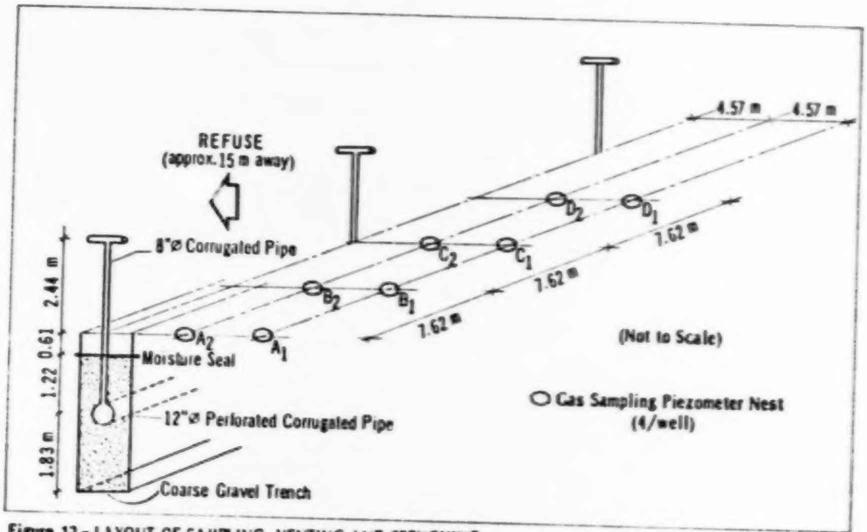
REFUSE
(approx. 15 m away)
8"⌀ Corrugated Pipe
4.57 m
4.57 m
D2
D1
C2
C1
B2
B1
A2
A1
7.62 m
7.62 m
7.62 m
2.44 m
0.61
1.22
1.83 m
Moisture Seal
12"⌀ Perforated Corrugated Pipe
Coarse Gravel Trench
(Not to Scale)
○ Gas Sampling Piezometer Nest (4/well)

Figure 12 - LAYOUT OF SAMPLING, VENTING AND TRENCHING SYSTEM FOR PREVIOUS MIGRATION STUDY

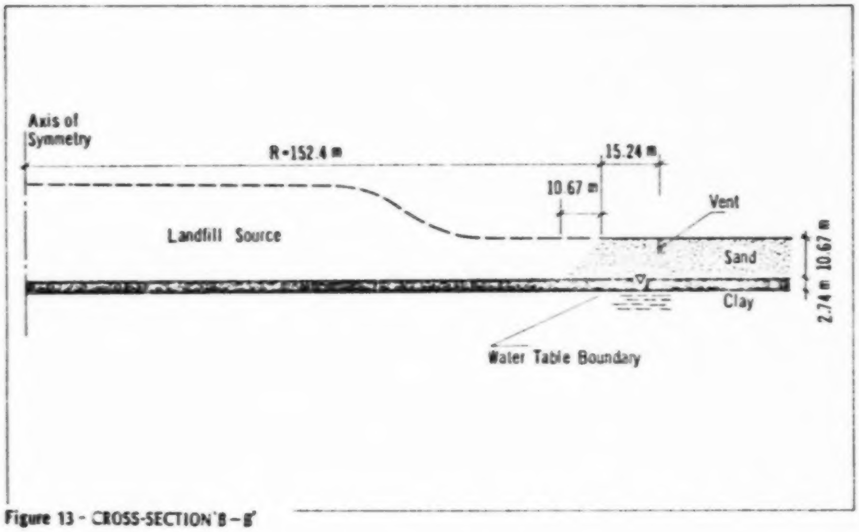
Axis of Symmetry
R = 152.4 m
15.24 m
10.67 m
Vent
Landfill Source
Sand
10.67 m
2.74 m
Clay
Water Table Boundary

Figure 13 - CROSS-SECTION B - B'

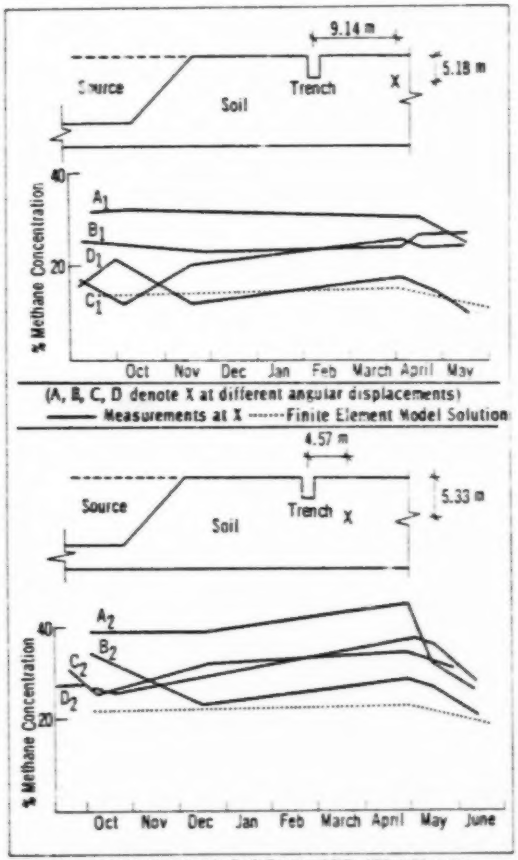

Figure 14 - COMPARISON OF GAS FLUX MODEL RESULTS WITH MEASUREMENTS

The results of the simulation as compared to the field measurements are shown in Figure 14, for methane concentrations at two points in the medium beyond the trench. Although the comparision is only fair, it must be noted that concentrations at sampling points A to D, which are at the same radial distance from the source as the simulated point, displayed substantial variability. The field data show that the trench as installed is ineffective in preventing the transport of methane gas. The model simulation presents the same conculsion. Consequently, use of the model in dèsign considerations would appear justified.

6. SUMMARY

A research program is currently underway to study pressures and gas compositions both in a landfill and at its periphery, and to use the results to calibrate a gas flux model.

Estimates of gas generation rates will be obtained by monitoring gas compositions and pressures before,during and after application of a zone of negative pressure within the landfill. This zone is provided by means of a gas withdrawal well installed vertically in the landfill and vented to a pump. A trial pumping test has indicated that sufficient drawdown is achieved throughout the monitored area to permit satisfactory measurements to be obtained, and has provided support for implementation of a theoretical technique for evaluating production rates using data collected during the recovery period after pumping ceases.

Several sets of pressure and concentration measurements have been obtained at the migration site. The influence and effectiveness of a forced venting system in operation at this site were apparent from these measurements. No methane was found in the area beyond the venting systyem on any of the study days. The data from all migration studies will be used to examine seasonal trends in pressure and methane concentration patterns, and to calibrate a gas transport model.

An example of the ability of the gas transport model to simulate field conditions was given. The model will be used in this study to display variations in gas migration patterns under a variety of soil conditions, and to evaluate various gas control methods.

The influence of atmospheric pressure on gage pressure readings was demonstrated. It is suggested that barometric pressures in the field be obtained at the same time as gage pressures, and that all pressures by converted to absolute pressures for purposes of comparison.

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